Cumulative Environmental Assessment in Tasmania: A Catchment Case Study

by

Andrew Harvey BSc.

University of Tasmania

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Declaration

This thesis contains no material which has been accepted for the award of any other degree or diploma in any tertiary institution, and to the best of my knowledge and belief, contains no material previously published or written by another person, except where due reference is made in the text of the thesis.

Signed

Andrew Harvey BSc.

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This thesis is an uncorrected text as submitted for examination.

Abstract

The cumulative impact of multiple stressors on the environment over time and space has been acknowledged as a key process in the ongoing loss of habitat and biodiversity and as a driver of landscape change. Regulatory approaches that address environmental impacts only at the level of the individual project footprint may facilitate significant environmental impacts through the cumulative effects of those projects. Such approaches are unlikely to ensure that current and future development of natural resources is sustainable. The catchment spatial scale and the regulatory processes that govern three key activities within it - farm dams, forest practices and water abstraction - form the basis for an examination of cumulative effects in Tasmania. International and Commonwealth regulatory approaches to cumulative effects, key concepts and methodologies are examined through a literature review. The potential cumulative impact of farm dams, forest practices and water abstraction on the natural flow regime, freshwater ecological processes and biota is established through the relevant literature. A case study of the Great Forester – Brid catchment in north east Tasmania is used to determine the potential for these impacts to occur in Tasmanian catchments. The results of this study show that there are measurable cumulative impacts on the natural flow regime, connectivity and special natural values within the catchment. Relevant legislation, policies and processes are examined to establish an understanding of how cumulative effects are addressed in Tasmania for these activities. Cumulative impacts of these activities are not adequately addressed in the current legislative and policy environment in Tasmania. An explicit legislative requirement for cumulative effects needs to be considered. In addition a pro-active, regional framework to manage and assess cumulative effects, incorporating integrated catchment management, is a fundamental requirement for the sustainable use of natural resources in Tasmania.

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Chapter 1 Introduction

Human societies impact on the natural environment through the imposition of multiple stressors. These stressors vary over time and across spatial scales. Regulatory, policy and management measures have been developed in response, as awareness of the need to ensure sustainable use of natural resources has increased. Despite this, there continues to be a measurable decline in biodiversity, an increase in habitat loss and fragmentation across the landscape and a continuing decline in the physical environment. One phenomenon in particular that has been identified as a key process in the continued decline of natural systems has entered the general lexicon in the form of the expression 'death by a thousand cuts'. This refers to incremental change through a multitude of apparently insignificant impacts which ultimately produce a cumulative effect greater than the sum of its parts. The expression 'tyranny of small decisions', first used in an environmental sense by Odum (1982), is a more appropriate expression for the consideration of cumulative environmental impacts that occur through multiple regulatory decisions which is the focus of this thesis.

Cumulative Environmental Assessment (CEA) is an attempt to identify multiple stressors at a variety of temporal and spatial scales and account for and manage their measurable impact on the environment. CEA is seen as an increasingly important practice in ensuring sustainable development. Although a relatively new field, there is none the less a recognised need for its application and a considerable body of work exists that provides methods, frameworks and analysis of issues associated with its application. The development of CEA practice has both arisen from and driven regulatory consideration of cumulative effects in a number of jurisdictions throughout the world. This thesis considers the current state of CEA practice and regulation in key international jurisdictions as well as in Australia. It examines the potential for cumulative effects to occur in Tasmania through common regulatory decisions at the catchment scale. The potential for implementing CEA in Tasmania within this context is assessed in terms of the current legislative and policy environment and the resources required for the implementation of appropriate CEA methodologies.

1.1 Definition of Cumulative Effects and CEA

A number of definitions of cumulative effects and CEA have been put forward. Definitions are provided in three key cumulative assessment guidance documents associated with regulation. The Council of Environmental Quality (CEQ) (1997, v) defines cumulative effects as 'the impact on the environment which results from the incremental impact of the action when added to other past, present, and reasonably foreseeable future actions regardless of what agency (Federal or non-Federal) or person undertakes such other actions' The reference to past, present and future impacts is an important feature of the consideration of cumulative effects, the use of the term 'agency' reflects the emphasis of US federal environmental legislation. The Canadian Environmental Assessment Agency (CEAA) (CEAA 1994, section 2) uses the following definition: 'The effect on the environment which results from effects of a project when combined with those of other past, existing and imminent projects and activities. These may occur over a certain period of time and distance'. The use of 'project' implies a more narrow definition than the use of the term 'action'. Hyder (1999, ix) offers a similar definition: 'Impacts that result from incremental changes caused by other past, present or reasonably foreseeable actions together with the project'.

Canter (1999, 406) provides a number of definitions from various authors. The use of terms such as 'valued environmental components' and 'project' used by some authors again seem potentially limiting, with the following definition from Rees (1995) broader in nature: 'cumulative impacts are the gross (or net) environmental impacts of a number of unrelated projects or activities (ie. multiple, qualitatively different impacts from a variety of causes and the interactions of these impacts) under conditions that result in time-or space-crowding'.

Finally, the following description from Finlayson et al. (2008, 1) is particularly applicable in the context of cumulative impact through multiple regulatory decisions: 'cumulative impacts occur where decisions made, apparently within the terms of legislation or prevailing policy, produce a series of small scale, individual outcomes that cumulatively have an effect contrary to the original intent of the legislation'.

Definitions of CEA are less problematic with the simple definition provided by Dube (2003, 724) almost self evident: 'cumulative environmental effects assessment (CEA) is the process of systematically analysing cumulative environmental change'. For the purposes of this thesis, the terms 'cumulative effects assessment' and 'cumulative environmental assessment' are considered interchangeable and are referred throughout as CEA. A more inclusive definition is provided in Canter (1999, 406) from Court et al. (1994): 'CEA involves predicting and assessing likely existing, past and reasonably foreseeable future effects on the environment arising from perturbations which are time- and/or space crowded, synergisms, indirect, or constitute nibbling'. 'Nibbling' is taken to mean small changes from multiple similar actions (Canter 1999, 407).

1.2 Nature of Cumulative Effects

Spaling and Smit (1993) suggest a consensus view exists for the characterisation of cumulative environmental change in terms of three attributes; temporal accumulation, spatial accumulation and the nature of human based activities. Temporal accumulation occurs where the interval between perturbations is less than the time required for the environment to recover from each perturbation with the rate of accumulation continuous, periodic or irregular. Spatial accumulation is where the spatial proximity between perturbations is smaller than the distance required for the effect of that perturbation to be absorbed by the environment and can be characterised by density, scale and configuration. Provided spatial or temporal accumulation occurs, the accumulation of environmental change will also be affected by the nature of anthropogenic activities.

The Canadian Environmental Assessment Research Council (CEARC 1988, 3) provides the following cumulative environmental effects typology, also adopted essentially in the same form by CEQ (1997):

- Time crowding: frequent and repetitive impacts on a single environmental medium;
- Space crowding: high density of impacts on a single environmental medium;
- Compounding effects: synergistic effects arising from multiple sources on a single environmental medium;
- Time lags: long delays before experiencing impact;
- Extended boundaries: impact distant from source;
- Triggers and thresholds: disruptions to ecological processes that result in fundamental change;
- Indirect effects: secondary impacts resulting from a primary activity; and
- Patchiness effects: fragmentation of ecosystems.

Cocklin et al. (1992) recognise two broad categories of cumulative impact. The first is the accumulation of impacts from multiple disparate impacts on environmental systems and the second is the accumulated impact on a single environmental component derived from multiple sources. Two pathways of accumulation are also identified. The first consists of impacts on various environmental components that remain disjunct. The second are processes that combine impacts additively, synergistically or through compounding (e.g. bioaccumulation). Figure 1.1 is a simple representation of cumulative effects pathways from Peterson et al. (1987). Another type of cumulative effect are those derived from indirect or secondary effects. Cocklin et al. (1992a) provides the example of a marina resulting in increased marine traffic, a more common example is road construction increasing the attractiveness of an area for urban or industrial development. These flow on impacts are described in Hegmann et al. (1999) as 'reasonably foreseeable actions'.

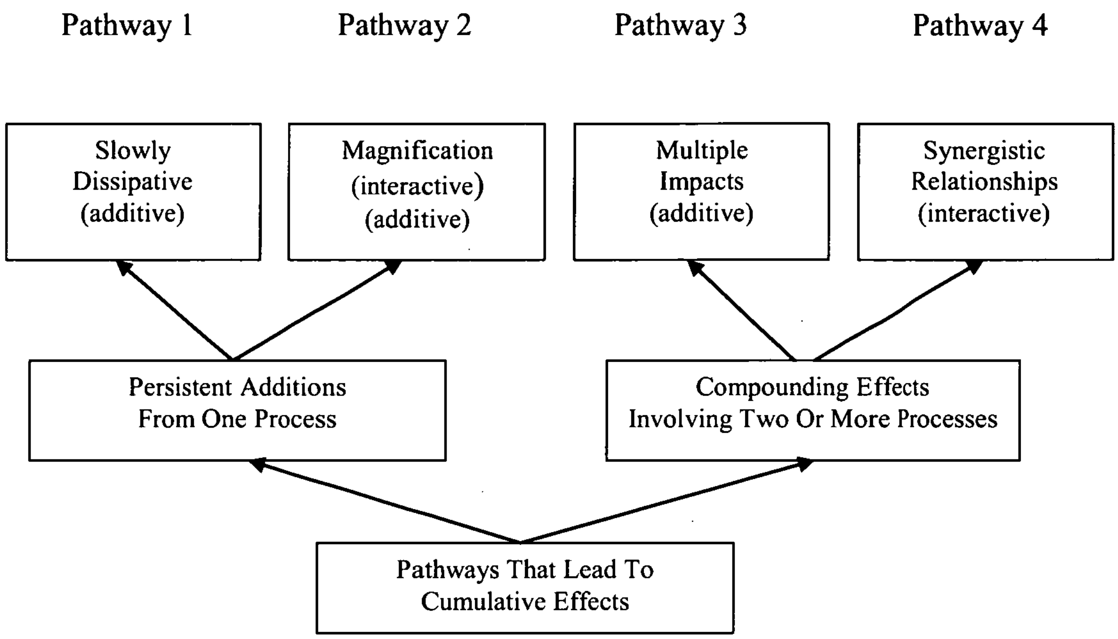


Figure 1.1 Basic functional pathways that contribute to cumulative effects. Source: Peterson et al. 1987.

Barrow (1997, 156) uses a categorisation of cumulative effects with a greater emphasis on accumulation pathways:

- Incremental (repeated additions of a similar nature);
- Interactive processes (a number of different impacts results in a significant impact);
- Sequential effects;
- Complex causation;
- Synergistic impacts;
- Impact occurs following exceedance of a threshold;
- Irregular surprise effects; and
- Impacts triggered by a feedback process.

Additive effects cannot be assumed to result in a linear response, and the crossing of a threshold resulting in a catastrophic or irreversible effect is a particularly important example of a non-linear response to accumulation of effects (Cocklin et al. 1992a). Non-linear responses may also occur through synergistic effects where multiple inputs interact to produce an effect greater than the sum of the inputs.

1.3 Need for CEA

The imperative for examining and managing cumulative effects arose out of the increasing perception from environmental assessment practitioners, governments and the community that environmental impacts were continuing to occur despite the application of EIA and the regulation that required it. In particular, landscape scale impacts and unmanaged landscape scale change were increasingly seen as areas of concern. The importance of cumulative effects arose through evidence of continued environmental degradation and a conceptual understanding of the nature of cumulative effects and their impact on the environment, which may often exceed that attributed to any single project (Canter and Ross 2008). As the practice of EIA progressed, its shortcomings also became more apparent, particularly in terms of its spatial and temporal constraints and also its reactive nature (Spaling and Smit 1993). It was also increasingly recognised that many activities that produce cumulative effects occur outside of the regulatory system, certainly outside of the project orientated EIA approach (CEARC 1988).

One of the key issues arising from a reliance on project EIA as the mechanism to assess and manage impacts of human activities is that it is largely a reactive process. The nature of proposals is often determined prior to submission for assessment which makes it difficult to fully consider all possible alternative uses either for a particular site or for a particular resource. CEA is seen by a number of authors as the foundation for a more proactive, regional approach to resource management. Such an approach is seen not only as the most effective method for managing cumulative effects but also to allow for better long term environmental planning through the control of conservation and development priorities (Gunn and Noble 2009, Dube 2003, Duinker and Greig 2006).

Consequently it has been recognised that without due examination and management of cumulative effects, the environmental sustainability of human activities cannot be determined (DEAT 2004). This is a view shared by the CEQ (1997) which also considers CEA as essential to achieving sustainable development and goes as far as to state that it is an impossible aspiration without it. Duinker and Greig (2006) argue for the transformation of EIA into a process that adequately considers cumulative effects in order to ensure the sustainability of Valued Ecosystem Components (VECs) in the face of human activities, which the authors describe as the central promise of CEA. The importance of CEA is perhaps most self evidently seen in the volume of effort apparent in its study, application and regulation. It is simply stated by Hyder (1999), that, as the effects of cumulative impacts can be significant, their management is necessary to ensure good decision making in the promotion of sustainable development.

1.4 Aims and Objectives

1.4.1 Aims

The aim of this thesis is to demonstrate the potential for cumulative effects to occur on a catchment scale in Tasmania through the regulation of farm dams, water licenses and forest practices and to assess the capacity of current legislation, policy and resources in terms of the assessment and management of those effects.

1.4.2 Objectives

The thesis attempts to achieve the following objectives in order to realise the stated aim:

a) To establish the need for the assessment and management of cumulative effects.

b) To review the current state of regulatory and methodological approach's to cumulative impact assessment.
c) To demonstrate the potential for cumulative effects to occur in Tasmania on a catchment scale under current regulation and policy through an examination of the farm dam approval process, forest practices system the water licensing process.
d) To identify legislative and policy opportunities and impediments for the adoption of CEA in Tasmania and to identify key resources that could be utilised for the application of CEA in Tasmania in this context.
e) To make recommendations for the adoption and application of CEA in Tasmania in relation to the farm dam approval process, water licensing and the forest practices system.

1.5 Methodology

A literature review is the basis for establishing the significance of cumulative effects in managing the human impact on the environment. In particular the review underpins the assertion that without proper consideration and management of cumulative effects the notion of sustainable use of natural resources remains unobtainable.

The literature review establishes the key regulatory approaches to cumulative effects across a number of significant international jurisdictions. Approaches, frameworks and methods in CEA are discussed. The review reveals the effectiveness and difficulties in both regulatory and methodological approaches to CEA.

To demonstrate the potential for cumulative impacts to occur on a landscape scale in Tasmania through regulated activities, the construction of farm dams, water extraction and forestry activities are examined at a catchment scale. Site specific and cumulative impacts of these activities on key physical and ecological indicators are examined through the literature. A case study of the Great Forester – Brid catchment is the basis for an examination of the potential for cumulative effects to occur through the regulation of these activities through the collation and analysis of data collected in the regulatory process and through spatial analysis.

Particular attention is given to the potential impact of farm dams in the catchment through the following approaches:

- Farm dam approvals are quantified, examined over time and the number of unregulated farm dams is estimated;
- The degree of fragmentation caused by in stream farm dams is analysed;
- The impact of farm dams on selected VECs is examined;
- The spatial distribution of farm dams within the landscape is examined;
- The CFEV database is used to provide an assessment of the cumulative impact on the modeled conservation values of river sections and wetlands; and
- The impact of farm dams, extraction and forestry on catchment yield is examined.

In addition, the spatial distribution of water licenses, certified Forest Practices Plans (FPPs) and plantation forestry is also examined.

For each of these activities the relevant legislation, policies and processes that govern them are examined and an analysis in terms of the adoption of CEA is undertaken.

Chapter 2 Literature Review

2.1 Introduction

This literature review examines key regulatory approaches to CEA, the nexus between CEA and Strategic Environmental Assessment (SEA) and regional assessment, key conceptual and methodological approaches to CEA and the difficulties and promise of conducting CEA. It will form the basis for examining cumulative effects within the Tasmanian context as it will enable a discussion of applicable CEA methods, resources requirements, regulatory effectiveness and strategic approaches to managing the cumulative effects identified in the catchment case studies.

Cumulative effects have been recognised as an issue of environmental concern since the latter part of the 1970s, although it is considered that attention to cumulative effects did not occur more thoroughly until the 1990s (Connelly 2008). The first attempt at addressing cumulative effects through regulation was in the United States in 1979, with Canada (1995) and the European Union (1985) also subsequently enacting significant regulatory measures. While other jurisdictions, for example South Africa (DEAT 2004), address cumulative impacts, the review is limited to these jurisdictions, with the addition of Australia, as they remain the most significant efforts to date and the associated guidance documents are widely recognised in the literature.

While regulation may provide an impetus to address a known issue, it may be limited in establishing accepted best practice in the methods employed to examine it. This limitation arises from the inherent nature of legislation and the difficulty in amending legislation to respond to changing social, political and environmental imperatives. There is often a disjunction between regulation and evolving practice and theory.

For CEA, a complex and evolving arena, there is a considerable body of work examining conceptual approaches, case studies and methods. Fundamental principles have been well established. Key methods have been developed, applied and reviewed. It is evident, however, that CEA practice has not fully met its promise, with the key debates currently focused on the evolution of CEA away from the initial regulatory requirement of addressing CEA in project EIA to a more proactive and strategic approach.

2.2 Regulation

2.2.1 United States

The primary federal environmental statute in the United States is the National Environmental Policy Act (NEPA). Enacted in 1969, NEPA effectively declares the United States' environmental policy (Clark and Richards 1999). Title I of NEPA contains a National Environmental Policy which requires the federal government to use all practicable means to 'create and maintain conditions under which man and nature can exist in productive harmony' (US EPA, n.d.). Section 102 requires federal agencies to incorporate environmental considerations in their planning and decision-making through a systematic interdisciplinary approach. Under this section, federal agencies are required to conduct an environmental assessment of all programs and actions. This includes policies, regulations and law (Lein 2003). Under Section 102(2)(c), where an action is considered likely to impact on the quality of the human environment, an Environmental Impact Statement is required. Private projects may also be subject to NEPA, where federal funding, assistance or permitting is required (Clark and Richards 1999).

Title II of NEPA created the Council on Environmental Quality (CEQ). The CEQ is an advisory body whose main role is to integrate environmental, social and economic actions within agencies of the federal government and develop and promote national policies. A key role is the oversight of the implementation of Section 102(2) (c)

(Clark and Richards 1999). The term "cumulative effects" first occurred in guidelines issued by the CEQ in 1973 (Canter and Ross 2008). The CEQ subsequently promulgated a number of regulations including the requirement that EIS address 'the incremental impact of the action when added to other past, present and reasonably foreseeable future actions, regardless of what agency (Federal or non-Federal) or person undertakes such other action. Cumulative impacts can result from individually minor but collectively significant actions taking place over a period of time' (Connelly 2008, 2).

In 1997 the CEQ released guidelines for addressing cumulative effects in the NEPA process (CEQ 1997). The guidelines addressed a gap that had resulted in federal agencies developing independent procedures for assessing cumulative effects. The guidelines outline key fundamental aspects for considering cumulative impacts and provide a range of methods and tools for conducting CEA. The guidelines require cumulative effects to be addressed over larger areas, longer time frames and the contributions from past, present and reasonable foreseeable future actions (Canter and Atkinson 2008). Although directed at the formulation of EIS under NEPA, the guidelines are an important reference for CEA practice in general.

The CEQ guidelines states that: '[t]he range of actions that must be considered includes not only the project proposal but all connected and similar actions that could contribute to cumulative effects', including future actions (CEQ 1997, 1). The Guidelines refer to the process of determining 'reasonably foreseeable future actions' to the individual entity responsible for conducting the CEA but advocates a defensible screening process for assessing projects at various assessment levels and progress and any possible impact. However, litigation has established that 'reasonable forecasting' is implicit in NEPA and the prediction of the environmental effects of proposed actions before they are known is a requirement (CEQ 1997). To overcome any uncertainty in this regard the use of scientifically accepted theoretical approaches or research methodologies is advocated.

2.2.2 Canada

The Canadian Environmental Assessment Act (CEAA), enacted in 1992, is similar in nature to NEPA in that it requires federal agencies to conduct environmental assessments of programs and policies prior to gaining federal funding (Western Economic Diversification Canada, n.d). The Act also applies when a federal agency is a decision making body for a private or public proposal, or where a federal agency is either the proponent, provides financial assistance, is involved in transactions relating to federal lands or issues licenses, permits or other authorisations (Clark and Richards 1999).

Subsection 16(1) of the CEAA requires every environmental assessment to include 'any cumulative environmental effects that are likely to result from the project in combination with other projects or activities that have been or will be carried out' (CEAA 2007), although the CEAA does not provide a definition of cumulative effect. Two extensive guidelines for the practice of cumulative effects under the CEAA have been released; *Cumulative Effects Practioners Guide* (Hegmann et al. 1999) and *Reference Guide: Addressing Cumulative Environmental Effects* (CEAA1994). Amendments to the CEAA, enacted in October 2003, recognise the use of regional studies as a tool in the consideration of cumulative environmental effects (CEAA 2007). Under Section 16.2, the results of a study of possible future environmental effects of possible future projects in a region may be taken into account, particularly in project level CEA.

Despite the lack of formal definition of cumulative effects under the CEAA, the key guidelines provide both a definition and guidance on key aspects of CEA such as anticipated future actions. The following definition is provided in the 1994 Reference Guide: 'The effect on the environment which results from effects of a project when combined with those of other past, existing and imminent projects and activities. These may occur over a certain period of time and distance' (CEAA 1994, 2). The Reference Guide indicates that CEA should be limited to only those environmental effects of the project that will accumulate or interact with those of other projects and activities. The CEAA requires that CEA considers those projects that 'will be carried out' (CEAA 1994). The Reference Guide suggests that as a minimum this would include all approved projects but that projects currently in an approval process should also be considered. Activities that are not subject to approval should also be considered where their occurrence is considered to have a 'high level of certainty' (CEAA 1994).

2.2.3 European Union

The European Union, established through the Treaty of Rome, the Single European Act 1986 and the 1992 Treaty on European Union (Maastricht Treaty), does not have a single environmental law that applies throughout member states but does seek to establish uniformity of standards through the use of Regulations and Directives (Bond and Wathern 1999). In the case of environmental standards, these have largely been implemented through Directives, which place obligations on member states to enact into the standards into law within set time frames, although individual nations may use them as a minimum standard. The majority vote required to pass Directives may result in these standards being correctly seen as such (Bond and Wathern 1999).

The 1985 Directive (85/337/EEC) applied to both private and public projects and provided a set of obligations for EIA at the project level, which required consideration of cumulative impacts (Marsden et al. 2010) through the use of the terms 'direct and indirect impact' and the 'interaction between factors' (Council of European Communities 1985). Subsequent Directives (97/11/EEC and 2003/35/EC) increased the scope of EIA and public participation in EIA respectively, with the former requiring consideration of the 'cumulation of projects' (Piper 2001). A further directive (2001/42/EC) established the obligation for member states to conduct SEA on plans and programs including cumulative and synergistic effects. The Habitats Directive (92/43/EEC) also requires consideration of plans and projects 'alone or in combination with others' (Masden et al. 2010). A guidance document for the examination of cumulative effects (Hyder 1999) was released by the European Commission in 1999. The guide largely addresses cumulative impacts in the context of project level EIA.

2.2.4 Australia

The principle federal environmental legislation in Australia is the *Environment Protection and Biodiversity Conservation Act 1999* (EPBC). The EPBC requires consideration of the protection and management of 'matters of environmental significance', which are defined under the Act (Department of Environment, Water, Heritage and the Arts, n.d.a). There are eight matters of national environmental significance; world heritage sites, national heritage places, wetlands of international importance, nationally threatened species and ecological communities, migratory species, Commonwealth marine areas, the Great Barrier Reef Marine Park and nuclear actions. The EPBC may also apply to other matters where either Commonwealth agencies of lands are involved. Under Section 142 of the EPBC the Minister may choose to assess the impact of a plan, policy or program through the use of a strategic assessment, although as it is at the discretion of the Minister this capacity of the EPBC is not considered to be able to be applied systematically (Marsden 1999). Such an approach may also be applied to areas that would otherwise be subject to a number of EPBC assessments (Department of Environment, Water, Heritage and the Arts, n.d.b).

Despite a clearly identified need for the inclusion of CEA in environmental decision making in Australia (Court et. al. 1994), CEA is not a legal requirement of the EPBC (Marsden 2002). Litigation, however, has made it clear that a wide consideration of impact, including cumulative impacts is required, with the Federal Court ruling that 'the Minister of the Environment must give the widest possible consideration to any project under the Act, having regard to the sensitivity, value and quality of the environment which is impacted, and upon the intensity, duration, magnitude and geographic extent of the impacts, including its whole, cumulated and continuing effect' (Connelly 2008, 2). However, while the significant impact guidelines refer to considering a project 'in the broadest possible scope', cumulative impacts are not specifically referred to (Commonwealth of Australia 2009). In a recent review of the EBPC (Hawke 2009), a strong theme in the submissions received was the limited ability of the EPBC to consider cumulative impacts and therefore to ensure sustainable development, particularly at the landscape and regional ecosystem level. The review, however, considers that cumulative impacts can be addressed in strategic assessment and through bioregional plans. It is also recognised that while the Minister can consider cumulative effects where the context of individual assessment would appear to demand it, there is no guidance policy to provide certainty or transparency.

2.3 Effectiveness of CEA Regulation and Practice

Despite the long term inclusion of CEA as a requirement of environmental regulation in some jurisdictions, the practice of CEA is considered not to have been undertaken to its full potential. CEA is considered to only to have been marginally considered in the EIA process in the United States (Canter and Kamath 1995) and within EIA practice in general (Canter and Ross 2008). A number of reviews of documents prepared for projects under NEPA found that CEA was either not addressed or poorly addressed with the overall proportion of documents that provided evidence of CEA low (Ma et al. 2009). Reviews in the United Kingdom and South Africa have found a similarly poor treatment of CEA in EIA (Marsden et. al 2010, DEAT 2004, Cooper and Sheate 2002). Duinker and Greig (2006) assert that CEA in Canada has not lived up to its promise and suggests that current practice may even be detrimental. Key deficiencies are identified at the project EIA level and in a lack of understanding of some key issues and poor interpretation of cumulative effects concepts. The added complexity involved in CEA methodologies have been summarised by Wijayanto (2002, 19) as arising from the need to;

- Determine the impacts that should be pursued;

- Bound the analysis in the spatial and temporal dimension and use this dimensions for an assessment of significance;
- Determine appropriate indicators;
- Determine appropriate methods and techniques to overcome data deficiencies; and
- Aggregate impacts to determine impact magnitude and significance.

Poor or absent treatment of CEA in EIA has been attributed to technical difficulties associated with the conduct of CEA, lack of institutional mechanisms and support, limited methodologies and the lack of a uniform methodological approach, and temporal and financial constraints (Ma 2009, Canter and Kamath 1995, Piper 2001, Piper 2002). One particular difficulty in the practice of CEA is the consideration of future actions, often referred to as reasonable foreseeable future actions (RFFAs) and this aspect of CEA has been the subject of considerable legal action in the United States (Rumrill and Canter 1997, Smith 2006). It has been suggested that this could be seen as a reflection of a generally poor understanding of CEA within EIA practitioners (Duinker and Greig 2006). The deficiency of capacity in this regard may also be seen from an analysis of CEA in Canada and the UK that identified that the fundamental issue of scale was often poorly addressed (Thrivel and Ross 2007). As a result of these apparent difficulties, Canter and Ross (2008) identify a lack of professional attention given to CEA. In particular, brevity of attention to CEA, lack of rigor, lack of systemic processes and documentation are highlighted.

There may also be a lack of commitment to good practice in the private sector and government agencies, a tendency to use uncertainty as a justification for not undertaking CEA and that CEA may be given minimal attention due to concerns related to funding of management and mitigation of cumulative effects. Another recognised key barrier to CEA practice is a lack of coordination between government agencies, multiple project proponents and scientific bodies (Masden et al. 2010, Ma et al. 2009) and jurisdictional conflicts (Canter 1999). Due to the added complexity of CEA, a gap has been identified between the regulatory obligations placed on a proponent and the requirements of a rigorous CEA (Piper 2001).

Despite the challenges and identified deficiencies in practice, opportunities are recognised for the continual improvement of CEA practice. Canter and Ross (2008) suggest that the relative newness of CEA allows for creative approaches, particularly in adapting existing management programs. Canter (1999) asserts that the view that there are insufficient methodological options available for CEA practice is erroneous, if consideration is given to modifying existing EIA methods. MacDonald (2001) suggests that the inherent challenge of CEA, along with the recognition and prediction of numerous direct and indirect effects, can be overcome through a better definition of resources of concern and spatial and temporal boundaries and the author presents an explicit process to achieve this outcome. The need for improvements in cross discipline approaches is also identified. A practical approach is used by the US EPA for wetland management, where cumulative effects are considered using a synoptic approach based on current data (Abbruzzese and Leibowitz 1997). This approach is applied for time critical decisions where the risk is low and therefore less rigour is deemed to be acceptable. Another successful practical application of CEA is illustrated by the CEA of the Northern River Basins in Canada, where stakeholder focused priorities were combined with experimental verification of causality and linkages between multiple stressors (Culp et al. 2000).

2.4 CEA in the EIA and SEA Processes

There has been an increasing focus in the previous two decades on the use of Strategic Environmental Assessment (SEA) in the assessment of plans, policies and programs (Canter and Reiger 2005). SEA is seen by some authors as representing a more suitable environment for the application of CEA (Canter and Reiger 2005, Bonnell and Storey 2000) as there is a greater opportunity to address multiple activities and actions across space and time although some considerable conceptual and methodological challenges have been identified (Gunn and Noble 2010). In addition, project level EIA is seen as having inherent limitations in its ability to analyze and assess cumulative environmental change (Canter 1999).

At the EIA level, temporal and spatial constraints operate on the analysis, with temporal boundaries generally that of the project lifetime, with emphasis on the initial implementation, and spatial boundaries typically bounded by local scales or jurisdictional limits (Canter 1999). These limitations often result in a narrow analysis of simple cause-effect relationships, first order impacts and an emphasis on the individual site in addition to limitations on the examination of baseline conditions at a broader scale (Canter 1999, Dube 2003). Proponents may also have limited information on other effects that may be cumulative (Dube 2003). Importantly the EIA process is usually triggered following the implementation of the regulatory process as it applies to the proposal and therefore pre-empts any anticipatory approach that may be more effective in managing cumulative impacts, particularly in regards to influencing initial design and justification (Canter 1999). Individual project EIA also has severe limitations on the proper consideration of the effects of other projects (Connelly 2008). Berube (2007) asserts that CEA should be separate component, with its own methodology. While there has been a response to these issues through the expansion of the scope of EIA, CEA has been viewed variously as a maturation of EIA, where CEA becomes the default EIA method, or as a planning tool, used, for example, in the conduct of regional natural resource assessment (Spaling and Smit 1993, Canter 1999, Dube 2003). Harriman and Gunn (2008) offer the alternative argument that views both EIA and SEA as suitable tiers of assessment for considering particular types of cumulative effects.

The development of SEA and regional assessment is in response to the need for an assessment of policies, plans and programs and other activities that may not be captured adequately by the EIA process. SEA generally has a larger geographic scope and longer timeframes and is considered therefore to have greater potential for consideration of cumulative effects, although the greater uncertainty requires greater innovation of methods and approaches (Canter and Reiger 2005, DEAT 2004, Gunn and Noble 2009a). Regional assessments focus on quantifying existing environmental effects and determining existing stressors (Dube 2003), which can feed into a specific SEA. In this way it may be possible to overcome some of the key difficulties of SEA identified by Therivel et al. (1994, 41), specifically the lack of information about existing and future environmental conditions and the nature, scale and location of future development proposals.

Regional assessments may also provide an important basis for the assessment of projects within the boundaries of that assessment or may indicate areas where particular projects may be considered inappropriate, thereby guiding proponents (Connelly 2008). They offer a focus on regional drivers of change, provide linkages through the tiers of assessment and are pro-active and future focussed (Gunn and Noble 2009a). The assessment of multiple stressors at regional scales can be used in both a retrospective and prospective manner and can provide a risk assessment framework for the consideration of cumulative effects (Gentile and Harwell 2001). Both SEA and regional assessment approaches can also identify requirements for project EIA and other forms of monitoring and review (DEAT 2004). The difficulty of regional assessments is that there is often not a legislative trigger, they often require multi-jurisdictional efforts over longer timeframes, require greater resources and are typically not ongoing (Dube 2003, Connelly 2008). Connelly (2008) advocates a legislative framework that links strategic or regional assessment to project assessment, publication of guidance for strategic and regional assessment and collaboration within government to address cost and time considerations.

Despite the acknowledged limitation of project EIA to adequately address cumulative effects and the apparent benefit of considering CEA through the SEA approach, Gunn and Noble (2010) argue that there has been little investigation into whether

CEA and SEA are well suited to one another, either conceptually or methodologically. Gunn and Noble recognise the limitation of project based CEA to predict and control the cumulative effect of human development actions and see SEA as providing an opportunity to set a future course of a region and pro-actively avoid the problems associated with individual decision making, including effects of development not subject to assessment. However, it is asserted that while the benefits of the SEA approach are well documented these benefits have not been fully demonstrated in practice. Key challenges are identified in the process of moving CEA beyond the project level through a survey of EIA/CEA practitioners are the establishment of an agreed definition of the nature of cumulative effects, aggregation beyond the scale of the individual project, the need to adopt a systems perspective, variability of approaches within SEA itself and differentiating between SEA and regional planning. Finally, Gunn and Noble emphasis another challenge that nonetheless also presents a potential benefit of SEA. This is the linking or 'tiering' between EIA and SEA, where one contributes to the other with SEA providing context to EIA and EIA responding to and contributing to SEA.

2.5 *Fundamental Principles of CEA*

2.5.1 Overview

Despite the debate surrounding the effectiveness of current CEA practice, the fundamental principles of CEA have generally been well documented. Key guidance documents have been produced (CEQ 1997, Hyder 1999, Hegmann et. al. 1999), however these largely focus on project level CEA. Further examination of the requirements of CEA in a broader sense has been well addressed in the wider literature and it should also be noted that fundamentals of EIA continue to apply in any CEA. The following examination provides an overview of some key considerations in the practice of CEA and is based on the following breakdown of components and steps provided the by Council of Environmental Quality (1997) that have been adapted and accepted by a number of authors (Canter 1999, DEAT 2004, Canter and Reiger 2005):

Scoping

1. Identify the significant cumulative effects issues associated with the proposed action and define the assessment goals.

2. Establish the geographic scope for the analysis.

3. Establish the time frame for the analysis.

4. Identify other actions affecting the resources, ecosystems, and human communities of concern.

Describing the affected environment

5. Characterize the resources, ecosystems, and human communities identified in scoping in terms of their response to changes and capacity to withstand stresses.

6. Characterize the stresses affecting these resources, ecosystems, and human communities and their relation to regulatory thresholds.

7. Develop baseline conditions for the resources, ecosystems, and human communities.

Determining the environmental consequences

8. Identify the important cause-and-effect relationships between human activities and resources, ecosystems, and human communities.

9. Determine the magnitude and significance of cumulative effects.

10. Modify and add alternatives to avoid, minimize, or mitigate significant cumulative effects.

11. Monitor the cumulative effects of the selected alternative and adapt management.

2.5.2 Scoping

Scoping is of particular importance in CEA practice as the scale of assessment will need to capture the full range of potential cumulative effects of significance while avoiding an unnecessarily complex and large study (Hegmann et. al.1999). At the project level this can be achieved by focusing on any effect to which the project may contribute and then examining wider cumulative effects on those components effected and is a combination of 'activity information' and 'environmental information' (Canter 1999). MacDonald (2000) suggests that the identification of environmental components of concern will largely determine spatial and temporal scope. Hegmann et. al. (1999) provide a number of steps in the scoping process including; issue identification, selection of environmental components of concern, setting of boundaries, identification of other actions and initial identification of potential impacts and effects. Environmental components should include those vulnerable to incremental effects or those effected by other similar projects, those effected by other activities in the area or where ecological processes may be altered (US EPA 1999).

Professional judgment is required to determine at what spatial scale the effects from a particular activity become insignificant and an adaptive approach is required following initial selection. Generally the spatial boundaries would be larger than that used for a project EIA (CEQ 1997). The CEQ (1997) suggest establishing a project impact area and expanding the study boundaries to fully include any significant components of interest. Boundaries should be scientifically defensible, for example boundaries may need to correspond to an appropriate natural range of terrestrial species or other ecological boundaries (US EPA 1999) or to fully encompass cause and effect relationships, and multiple boundaries may be required (Canter 1999). Hyder (1999) indicates a preference for the use of natural boundaries as opposed to administrative boundaries.

Temporal boundaries also require scoping and CEA, as it requires consideration of past and future actions, presents a particular challenge in this regard. Hegmann et al. (1999) suggest that the temporal boundaries should begin ideally before the effects of the action or other major actions are measurable. Dube and Munkittrick (2001) propose a CEA framework which uses an effects based assessment to establish the existing environmental state and a stressor based assessment to predict potential future impacts. Future considerations should ideally extend to the point where the effected ecosystem components return to the pre-action condition, although the longer timeframes required are likely to result in an increasing reliance on qualitative approaches. National or regional planning timeframes should also be accounted for in the selection of temporal boundaries (Hyder et al. 1999). Hyder et al. (1999) also suggest that time frames beyond five years have too much uncertainty, although this view is in the context of project CEA. More strategic studies must consider longer timeframes. Berube (2007) suggests that ten years is the furthest extent, based on the experience of twelve CEA for hydro developments in Canada. A SEA of a long term investment plan for navigation on the Ohio River had an analytical timeframe extending to 2060 (Canter and Reiger 2005). Despite the uncertainty, a range of temporal scoping considerations are well documented (Canter 1999), that, in combination, should provide a defensible temporal scope.

A related scoping component is the selection of future actions. Hegmann et al. (1999) characterises future actions into three categories depending on uncertainty; those that are considered highly likely to proceed, those that a reasonable likely to proceed, including actions likely to occur as a result of project approval, and those that are largely hypothetical. It should be noted that this categorisation is restricted to those actions dealt with by regulation. Spatial boundaries can also be used to identify potential future actions that may contribute to effects on selected components (CEQ 1999), while the selection of the VEC may in turn determine the consideration of RFFAs if they are likely to impact the VEC within the scope of the assessment (Canter and Ross 2008, US EPA 1999). The issue of RFFAs has been extensively considered in the US courts, and those deliberations have formed the basis of an eight step RFFA process intended to provide a methodical procedure (Canter 1999, Rumrill and Canter 1997), which is not provided in the guidelines issued by the CEQ (1997).

2.5.3 Analysis of Effects and Description of Environment

Hegmann et al. (1999) adopt an analysis of cumulative effects that focuses on VECs, with broad guidelines provided on the selection of appropriate tools to conduct such an analysis. VECs are seen as the pivotal focus of the assessment with the CEA conducted 'from the VEC point of view'. VECs are defined as 'any part of the environment that is considered important by the proponent, public, scientists and government involved in the assessment process. Importance may be determined on the basis of cultural values or scientific concern (Hegmann et al. 1999) and this is an important aspect as it allows for community input which may provide greater validity.

Hyder (1999) identify a number of key stages in the analysis of effects, again at the project level:

- identify where indirect and cumulative impacts and interactions will potentially occur;
- identify the cause and effect relationship – the pathway that impacts will follow which will show how project activities will impact on the existing environment;
- determine the response of the resource to a change in the environment, including assessing the magnitude and the significance of the impacts;
- developing mitigation measures to address the impacts; and
- developing monitoring programmes to gauge the indirect and cumulative

impacts, and impact interactions, and establishing mechanisms for addressing significant impacts if identified.

The magnitude of impacts should be quantified, where possible, or where a qualitative approach is necessary, ranked (Hyder 1999). The CEQ (1997) provide similar series of steps following confirmation of the resources and effects to be considered:

- identify the important cause and-effect relationships between human activities and resources, ecosystems, and human communities.
- determine the magnitude and significance of cumulative effects.
- modify or add alternatives to avoid, minimize, or mitigate significant cumulative effects.
- monitor the cumulative effects of the selected alternative and adapt management.

2.5.4 Significance of Effects

Determination of significance relies on both the establishment of probability and the spatial and temporal components of the effect, with a focus on the VEC, rather than the action (Canter and Ross 2008). Hegmann et al. (1999) consider effects on biological and physical-chemical VECs separately, and consider of the extent of the effect in relation to natural variability (physical-chemical) and reproductive capacity or habitat (biological), with both considerations addressing recovery rates and extent, and restoration to 'acceptable conditions'. The use of VEC thresholds is an important component of significance analysis, as it cannot be assumed that an incremental scale of significance can be applied. The determination of thresholds, however, may be problematic and may ultimately rest on professional judgment (Canter 1999, Canter 2000) although some may be regulated (US EPA 1999).

Scope can also influence determination of significance, and there is a risk that larger spatial and temporal scales may result in some effects appearing to be insignificant at that particular scale. Other determinations include: legislative, planning and policy considerations, public concerns and land use zonations (Canter 1999, Canter 2000). Finally, the effectiveness of mitigation of cumulative impacts must also be considered in the determination of significance (Canter and Ross 2008).

2.5.5 Methods and Approaches in CEA.

The following examination of CEA tools and methodologies focuses on a list of methods derived from the CEQ (1999), Canter (1999), DEAT (2004), Smit and Spaling (1995) and Canter and Ross (2008). The list of methods includes those methods that have been utilised in CEA or are considered to be suitable for adaptation in CEA. It is not inclusive of every method but does describe the key methods identified in CEA and demonstrates the variety of tools and methods available.

2.5.6 Questionnaires

Questionnaires, interviews and panels can be important techniques for analysing cumulative effects as they can gather a wide range of actions and effects and can consider social and cultural issues in addition to environmental concerns (CEQ 1997). A further advantage is that this method considers the impacts early in the CEA process (Hyder 1999). The method is contingent on the use of existing data to formulate the questionnaire and reconsultation may be required as the CEA progresses (Hyder 1999). This approach is often an important tool in the scoping process and can be used to prioritise the importance of cumulative effects. Panels, expert opinion and other group decision methods, although subjective, can utilise evaluation techniques to provide a ranking of importance (CEQ 1997). Canter and Reiger (2005) provide a case study from the Ohio River where a number of expert working groups provided continuous scoping and research priority input into large scale SEA. This technique has the advantage of flexibility and can deal with a wide range of subjective information, however it cannot provide a quantitative basis for decision making and comparison of alternatives is a subjective process (Canter 1999).

2.5.7 Checklists

A related, more systematic method than questionnaires, is the use of checklists. Checklists, by providing a list of common or likely effects, allow for the identification of potential effects and reduce the likelihood of effects being overlooked (CEQ 1997). However, while standardised checklists are repeatable, they may not be complete for every study, or can be unwieldy and lead to double counting. Checklists are not a substitute for thorough scoping, do not illuminate cause and effect relationships and are qualitative in nature (Canter 1999, MacDonald 2000). However some of problems of checklists can be overcome through the use of project or activity specific checklists or tiered checklists (CEQ 1999). A number of United States agencies have standard checklists for particular activities, while checklists can also be developed for specific components of CEA such as consideration of past, present and future activities (Hyder 1999). Canter and Kamanth (1995) devised a comprehensive checklist for scoping cumulative impacts in an attempt to provide a systematic approach, although the authors acknowledge the importance of combining the method with other scoping techniques.

2.5.8 Matrices

Matrices are essentially two dimensional checklists designed to assess the magnitude and importance of individual interactions between actions and resources that have been extended to apply to cumulative impacts (CEQ 1997). Matrices do not quantify effects but combine quantitative results to evaluate cumulative effects of multiple actions on resources, they can sum additive and interactive effects and identify higher order effects (Smit and Spaling 1995). Generally effects are scored according to magnitude, duration, probability or importance and values may be rankings or measurable quantities (CEQ 1997). Where weightings are used, this approach relies on expert opinion and introduces a complexity that may be difficult to interpret for third parties (Hyder 1999). However, for large, complex studies this is likely to be necessary. The Ohio River study referred to above utilised 22 matrices, all prepared and reviewed by a number of expert committees (Canter 2008). It may also be problematic to use weighted results in an additive way, for example to determine combined impact, as they may not be strictly additive.

Matrices can be extended to stepped matrices that display resources against other resources to facilitate tracing effects through the environment. Matrices do not directly address spatial and temporal scope as these need to be well defined to construct the matrix (Canter 2008). Matrices do not address cause and effect relationships but can address alternative and multiple projects (Canter 1999). However, the results of matrices can feed into considerations of spatial and temporal scope, for example through the use of matrices to consider RFFAs (Canter 2008).

2.5.9 Modeling

Modeling can be employed to quantify the cause and effect relationships leading to cumulative effects, either mathematically or through the use of expert systems (CEQ 1997). Developing specific models may require substantial resources and CEA usually utilises and modifies existing models, although in some instances the use of models may be constrained by limitations in project specific baseline data. The collection of specific data may be time consuming and the quality and context of the data are vital in the interpretation of results (Schneider et al. 2003) in addition to the accuracy of the simulations (Smit and Spaling 1995). Models often require a number of assumptions, which may be poorly understood by stakeholders, resulting in low acceptance. Despite this, the CEQ (1997) consider models to hold considerable promise. They can, provide an opportunity to quantify effect and resources mathematically. There is also a view that the development of models may facilitate communication between stakeholders (Schneider 2003).

Some aspects of the environment are particularly suitable for the use of modeling as an analytical tool including; air and water quality, hydrology, noise and airborne deposition (Hyder 1999). Simulation models that examine component processes within ecosystems can generally differentiate additive and interactive processes and may provide an analysis of pathways of environmental change. A key limitation in the use of models is often the lack of validation from observations over a range of site conditions, the difficulty of determining error sources in complex models (MacDonald 2000) and the requirement of a reasonably well understood system (Smit and Spaling 1995).

2.5.10 Networks

Network or system analysis is based on the concept of links and interaction pathways between individual environmental elements, where an effect on one element will be evident in those that interact with it (Hyder 1999). The analysis identifies the pathway of an impact using a series of network or system diagrams between an action and the receptor of an impact and is considered the most effective tool for examining cause and effect relationships (CEQ 1997). Analysing the receptor response and identifying effects on related receptors enables indirect impacts and interactions to be considered (Hyder 1999). Cumulative impacts can be identified where different actions impact the same receptor. Feedback can be incorporated to produce a loop or system analysis, although networks are hierarchical and therefore may not be able to consider all relationships (CEQ 1997).

The key advantage of this method is that it makes explicit multiple and complicated impacts, including indirect or secondary impacts, with the mechanism of cause and effect made clear. The method may not be quantified but can be used to identify processes that may require a quantitative approach. This analysis, however, does not provide an evaluation of cumulative effects (Hyder 1999). Where quantitative measures are used, the evaluation using a common unit of measure is a feature of system and network analysis, however different classes of effects requires separate evaluation (CEQ 1997). Network and system analysis cannot examine spatial and temporal components of CEA (Canter 1999, Smit and Spalding 1995). This weakness hampers the analysis of structural and functional change (Smit and Spalding 1995).

2.5.11 Trend analysis

Trend analysis is simply the assessment of the status of resource or an ecosystem component over time and as such provides an insight into the effects of past actions that may be then projected into the future (CEQ 1997). Trend data can be used to establish a baseline where current data may be insufficient and can identify historical cause and effect relationships. It can also be a prime method for identifying resources subject to potential cumulative losses by examining changes to intensity or occurrence of stressors over the same time period (CEQ 1997, Canter 1999). The establishment of a pattern through trend analysis can be a critical aspect of the identification of cumulative impacts (Hyder 1999). Trend analysis may be based on simple quantitative relationships or time series satellite imagery. The selection of habitats for trend analysis captures an important cumulative effect indicator.

2.5.12 Geographic Information Systems

Geographic Information Systems (GIS) can be used to examine the spatial distribution of environmental attributes, the change in that distribution over time and the correlation with land use or development patterns (Smit and Spalding 1995). Swenson and Ambrose (2007) identified substantial cumulative loss of wetland habitats through regulation using remote sensing, GIS and spatial analysis. While the use of GIS in this manner is a form of trend analysis, GIS has a wider application in CEA. GIS is an important tool in the identification of the spatial distribution of potential receptors, identification of physical cause and effect pathways, analysis of landscape parameters, determination of significance through spatial analysis of effects, and the overall spatial extent of the CEA (Canter 1999, Antunes et al. 2001, Cocklin et al. 1992b). The overlaying of a series of spatial layers, each depicting an effect from an action, will produce a cumulative effects map (CEQ 1997). GIS can also be used to plan for cumulative effects by examining the resources capability of an area, including key environmental constraints (Hyder 1999). It can provide a mechanism for examining future scenarios by mapping changes in environmental components in response to modeled effects (Atkinson et al. 2008, Kepner and Edmonds 2002)

The key weakness of GIS is that while cause and effect relationships may be inferred from spatial analysis, GIS does not identify or analyse those relationships (Smit and Spalding 1995). Additive and interactive processes are not differentiated and spatial changes of an environmental variable are assumed to be the result of the same process over time. Data requirements and data variability are likely to be key restraints.

2.5.13 Carrying Capacity Analysis (Thresholds)

Carrying capacity, or threshold analysis, is based on the understanding that natural thresholds exist in natural and man-made systems and that these limits can be applied to the consideration of the effects of cumulative impact (Hyder 1999). In ecological terms, the carrying capacity could be used as the level of environmental stress that a population or ecosystem could sustain without permanent damage, below which populations or ecosystem functions can be sustained (CEQ 1997). For carrying capacity analysis to be effective, appropriate limiting factors on a particular resource must be selected. Thresholds can be derived from expert opinion, survey, modeling or from regulation (Hyder 1999). As with modeling, carrying capacity analysis may be constrained by a lack of data at a regional level (Canter 1999).

2.6 Summary

There is little doubt regarding the need for conducting CEA. It is reflected in the formulation of regulation in the United States, Canada and the European Union that explicitly requires it. There is also a general acceptance, evident in the environmental assessment literature. It is apparent, however, that there are significant problems with the application and effectiveness of CEA in jurisdictions where it has been required by law. These difficulties largely arise from regulation that requires a process to be undertaken that is inherently complex and is relatively novel in its application. The attempt to address the variable response to regulation through the release of guidance documents only partially resolved the key difficulties.

There is an inherent limitation in assessing cumulative impacts through the project level EIA process as is required by the key regulatory mechanisms examined in this review. This is not to suggest that cumulative effects should not continue to be addressed in this context and the suggestion that CEA should be viewed as simply a maturation of EIA has merit. Improvement in the treatment of cumulative effects through project EIA can be achieved by addressing the issues identified in this review, particularly those related to capacity and understanding amongst practitioners. Further improvement could be expected as CEA methods continue to be reviewed and assessed from practice.

SEA is widely seen as a more appropriate vehicle for the assessment of cumulative effects primarily due to its expanded scope. SEA is also arguably a more preemptive, although not necessarily proactive, approach that allows consideration of cumulative effects to be fed into the resulting project EIA requirements. SEA itself, however, is an emerging practice and there is uncertainty and debate about approaches and methods. Indeed these issues mirror those identified in CEA practice as leading to poor outcomes. The current evolution of this approach is such that there are only a limited number of examples that can be critically assessed.

The greatest promise for the practice of CEA would seem to be the use of a regional CEA that is proactive in nature and would form the basis of actively directing the use of natural resources within a region. Such an approach avoids the reactive nature of project EIA, and arguably SEA to some extent, as it considers development scenarios and can potentially allow for control to be exercised over the type, temporal extent and spatial distribution of developments. In this context, development proposals to be considered through project level EIA could more readily examine cumulative effects because the methodological frameworks and relevant monitoring programs would be in place and the development itself would only be considered if it adhered to an overall strategic approach that would already have considered cumulative effects. In addition developers and the community would have greater surety regarding developments would be considered suitable for a particular region. The greatest difficulty in many jurisdictions with this approach is that there is often no regulatory or policy imperative for its initiation and ongoing implementation.

Chapter 3 Catchment Impacts: Farm Dams, Forestry and Water Extraction

3.1 Literature Review

3.1.1 Introduction

This chapter examines the cumulative and direct impacts of farm dams, forestry and water extraction. In the case of farm dams and water extraction, the focus is on the catchment spatial scale and the impact on ecosystems, species and processes related to catchment hydrology. Forestry may have cumulative impacts on terrestrial biodiversity, however for the purposes of this study these are confined to the impact on hydrology and associated ecological processes and dependent biodiversity. Each activity is examined as an individual stressor, although clearly a number of these impacts are common to all and in some way result in alteration of the natural flow regime.

The current understanding of the impact on catchment hydrology and catchment processes and ecosystems of farm dams, forestry and water extraction is reviewed. The aim of the review is to establish the current state of knowledge and to provide the contextual basis for an examination of a Tasmanian catchment, the Great Forester – Brid in the north east of the state.

3.1.2 Review of Known Impacts

3.1.2.1 Farm Dams

Site Specific Impacts

The impoundment of a stream or gully results in the clearance and conversion of both aquatic and terrestrial habitat within the dam footprint and area of inundation. The loss of permanent and ephemeral riverine habitat or wetlands to an impoundment reservoir with a highly variable capacity is likely to lead to the loss of the majority of instream fauna, flora and riparian vegetation. Terrestrial fauna of insufficient mobility or species unable to access connective habitat, for example burrowing crayfish, will be lost from the site. Riparian and other terrestrial vegetation within the area of impact is cleared and converted. This may include rare or threatened species. Terrestrial fauna associated with particular habitats will also be impacted, for example, a colony of the endangered Forty-spotted pardalote (*Pardalotus quadragintus*) was lost following the clearance of *Eucalyptus viminalis* forest, a key requirement of the species, for the construction of a farm dam (Bryant 2010). Some farm dams may be also sufficiently large as to lead to the loss of geoconservation features.

Catchment Yield

Farm dams are designed to capture and store runoff that would otherwise contribute to the natural hydrological regime of a catchment. Consequently one of the cumulative effects of farm dams within a catchment is to reduce catchment yield. The impact of the farm dams on catchment yield has been the subject of investigation in the Murray-Darling Basin, in Victoria and South Australia and to some degree in Western Australia. The impetus for this work has been the increasing awareness of the need to better manage water resources in south-eastern Australia and the recognition that farm dams can have a significant effect on catchment yield that requires quantification. Investigations into the impact of farm dams on catchment yield have produced estimates of between 3 and 50 percent of mean annual flow (Duggan et al. 2008).

An investigation of the effect of catchment farm dams (dams that are not on a waterway but have their own catchment) in Victoria, using an approach based on a water balance model (Tool for Estimating Dam Impacts (TEDI)), found that for every megalitre of storage capacity between 1 and 3 megalitres was lost from the annual downstream flow (Neal et al. 2000). A similar technique applied to the

Campapse River catchment in Victoria found that farm dams reduce annual catchment yield by 9%. Further investigations suggest the impact on stream flow in this catchment is likely to increase under climate change projections (Sinclair Knight Merz 2008). Overall, 90% of catchments across Victoria have a streamflow reduction of between 0.3 and 1.1 ML for every 1 ML of farm dam volume (van Dijk 2006). A similar result was found in a study of seven catchments in Western Australia (Sinclair Knight Merz 2007).

In South Australia, the level of farm dam development has been recognised as resulting in a reduction of median flows (South Australian Murray-Darling Basin Natural Resources Management Board 2006). In the Marne River catchment in the Mt Lofty Ranges a direct correspondence between farm dam capacity and streamflow reduction has been established (Nathan et al. 2000). For every megalitre of farm dam storage an equivalent volume is lost from annual streamflow. A separate study of the upper Marne catchment (Savadamuthu 2002) found that 640 farm dams reduced the annual adjusted runoff (generated under a scenario without the impact of dams) by 18% at the mean and 24% at the median. McMurray (2003) found that approximately half of the volume lost from farm dams could be attributed to evaporation. A modeled assessment of the South Para River catchment, north east of Adelaide, found that farm dams reduce the predevelopment median flow by approximately 7% (Teoh 2007). A similar study of the Onkaparinga catchment, south of Adelaide, found that farm dams reduced the median annual streamflow by between 5 and 8 % (Teoh 2002).

Within the Murray-Darling basin the cumulative impact of farm dams on streamflow is recognised as significant (van Dijk 2006, Schreider 1998), with the total impact on streamflow across the basin estimated at 1 900 GL per year. The projected future impact of farm dams to 2030 is predicted to lead to an additional decrease on runoff averaged across the basin of 0.65% but up to 10% in individual catchments (Jordan et al. 2008, Schreider et al. 2002). A study examining the impacts in greater detail within twelve individual catchments (one undeveloped catchment was used as a control) found a statistically significant reduction in streamflow following an average annual increase in farm dam capacity of 1.5% and 3.3% of mean annual flow (Schreider et al. 2002). In general, while the impact on total streamflow across the basin of future farm development may not be significant, the impact on streamflow within sub-catchments could be expected to be significant (CSIRO 2008).

Hydrology

The cumulative impact of farm dams on the hydrology of catchments cannot be understood simply in terms of reduction in catchment yield. The impact is variable between seasons and between years and the impact may also vary with flow independently of average seasonal flows. Farm dams can change the magnitude of flows and the timing and duration of flood peaks and flushes. Where farm dams are below capacity due to usage or evaporation, inflows are captured and not released downstream in each successive dam until it spills, thereby delaying and attenuating the natural flow.

This was found to be the case in the Victorian case studies of catchment dams (Neal et al. 2000) where they had the greatest impact on monthly low flows (flow exceeded more than 90%) than median monthly flows. This result was evident in a study of a small catchment by Neil and Srikanthan (1986) where the most significant reduction in catchment outflow occurred during the driest months, a finding replicated by Alcorn (2009) for the Eyre Peninsular. During wetter months when the dams were near capacity the impact was least and was found to be negligible in high flows due both to the overwhelming volume but also the increased likelihood that dams were near capacity prior to high flow events. The greatest impact was on late autumn and winter flows when the dams were at lowest capacity and on low flow spells where the cumulative effect of the dams was found to be an increase in the frequency, duration and variability of low flow spells.

The delay of autumn or early winter flows or other flow pulses throughout the year may impact on biological responses and is likely to lead to reduced water quality that may also effect aquatic biota (Dare et al. 2002, McMurray 2006). The reduction in flow duration, an increase in low flow through a reduction in peak flow events and a heightened seasonal impact at the onset of the higher rainfall periods of the year has been generally recognised as a consequence of the cumulative impact of farm dams (Good and McMurray 1997, South Australian Murray-Darling Basin Natural Resources Management Board 2006). These effects are more evident where a chain of dams occur (Duggan 2008). Similar patterns of impact in relation to seasonal flow variation and dam capacity were found throughout the Murray Darling Basin (Jordan et al. 2008, Chiew at el. 2008). Callow and Smettem (2009) found that in catchments dominated by dams the hydrograph was broader and flatter with lower peak flows and shorter flow duration.

Farm dam impacts will also vary between years in response to inter-annual rainfall variability. Teoh (2007) found the impact of farm dams on the median flow for the upper Para River catchment to be a 9% reduction but increasing to a 19% reduction in a dry year period, a result also found in a similar study on the Eyre Peninsular (Acorn 2009). This effect is also recognised by Teoh (2002) who determined that during low flow periods farm dams intercept a proportion of median flow equivalent to that captured on average by catchments with a dam density of over twice that of the studied catchment. McMurray (2006) found that farm dam development in the Tod catchment in South Australia was unlikely to have a significant impact in median and wet years but contributed to high levels of environmental stress across the catchment during dry years. A South African study of the cumulative effect of farm dams found that the impact was significant for baseflows, suggesting an increased impact during low rainfall years (Mantel et al. 2010a). While a consistent finding is that the effect of farm dams is least significant during high flows it is measurable. Kozarovski (1996) found that the volume of farm dams within the Torrens River catchment could lead to underestimation of peak flood flows with computed discharges greater than actually recorded due to unaccounted farm dam storage. Changes to low flows and peak flows has the potential to modify downstream riparian vegetation (Reid 1993).

A further aspect of variability within the cumulative impacts of farm dams on hydrology is the spatial distribution of dams within a catchment. A number of studies on the impact of farm dams focus on whole of catchment effects. However, a study into the increase in farm development across the Murray-Darling Basin (Murray-Darling Basin Commission 2008) recognises that the concentration of farm dams in particular areas lead to significant impacts in local catchments and the relationship between farm dam density, location within the catchment and impact has been documented in other studies (van Dijk et al. 2006, Davis 2003, Good and McMurray 1997, Teoh 2002, Mantel et al. 2010a).

Fragmentation and Connectivity

Hydrological connectivity is described by Pringle (2003, 2685) in an ecological context as the 'water mediated transfer of matter, energy and/or organisms within or between elements of the hydrological cycle' and it is considered essential to the ecological integrity of the landscape. Reduction or enhancement of this element can have significant environmental consequences. Hydrological connectivity can also be considered to be between watercourses and the surrounding landscape, with this relationship controlling the movement of water and sediment through the landscape (Callow and Smettern 2009, Allan 2004). Connectivity in riverscapes is also not limited to a single direction, the downstream direction of flow, but is also expressed as a series of downstream-upstream linkages (Pringle 1997).

The most recognised impact of dams on connectivity is the fragmentation of fish habitat through the creation both of a physical barrier but also the creation of potentially hostile habitat (Freeman et al. 2002). However the disruption of connectivity can have other less apparent consequences and downstream effects must also be a consideration (Merrill 2001). For example, the cumulative effect of dams may reduce the transport of inorganic dissolved solute silica, an important component of coastal food webs (Pringle 2003). Another measurable impact of impoundments is fragmentation of floristic communities containing species whose propagules have poor floating capacity (Jansson et al. 2000). Overall the unprecedented loss of hydrological connectivity across the globe is considered to have resulted in significant losses of biodiversity and ecosystem integrity.

In-stream farm dams have the effect of preventing fish movement upstream and of isolating upstream populations. Kashiwagi and Miranda (2009) found altered fish assemblages upstream of impoundments, attributed to the loss of connectivity with downstream reaches due to the cumulative effect of farm dams. Small impoundments were found to have significant effects on fish fauna at small geographical scales but also to have potential cumulative effects across catchments if not managed strategically. Alexandre and Almeida (2010) found similar results for upstream reaches above small instream obstacles including farm dams. Diadromous and migratory fish are particularly vulnerable (Merrill et al. 2001). The altered assemblages resulted from the barrier effect of impoundments prevented recolonisation of streams following drought. It also increases the risk of local extirpation above impoundments due to the inability of populations to move away from habitats diminished through reduced flow resulting from climatic and seasonal variation (Freeman et al. 2001). This effect has been recognised by Davis (2003) who suggests that species may tolerate some loss of connectivity but there are likely to be thresholds of fragmentation across a species habitat. Fish populations above barriers may survive for some time but remain at risk to stochastic events over the longer term (Santucci et al. 2005). The isolation of upstream populations can produce genetic and species level changes through reduced genetic flow and variation (Pringle 1997). In Tasmania, habitat degradation and hydrological alteration are identified by Hardie et al. (2006) as threatening processes for the majority of the islands *galaxiid* species.

Farm dams tend to be located predominantly on lower order streams in the mid to upper catchments of tributaries. Small headwater streams may represent up to 75% of total stream length within a catchment (Barmuta et al. 2009). Headwater streams are highly responsive to landuse and its effect on inputs of energy and material and are inextricably linked to the health of downstream reaches. Due to the proportion of stream length they have critical roles in the retention and breakdown of carbon, nutrient cycling and sediment transport (Barmuta et al. 2009, Freeman et al. 2007). Intact linkages between estuaries and headwaters are a critical component of ecosystem health (Davies 2003). There is an intrinsic link between hydrological processes and landscape coupling in headwater streams and downstream water quality (Alexander et al. 2007). Headwater streams often do not support permanent populations of aquatic biota, however they are known to support a higher proportion of narrow range or endemic species. In addition they have important roles as refugia (Freeman et al. 2007). With the exception of forested headwater streams there is a significant knowledge gap in relation to the functioning of headwater streams and the impact on their connectivity with downstream reaches through farm dam development may not be fully appreciated.

Habitat, Sediment, Water Quality and Biota

Farm dams can have significant downstream impacts on biota, either through an alteration to flow regime, habitat, nutrient cycling or water quality (Freeman et al. 2001). Not surprisingly, macroinvertebrate indices used for measuring river health are found to be higher in free flowing river sections as opposed to impoundments due to altered and degraded habitat and poor water quality (Santucci et al. 2005) However macroinvertebrate and fish assemblages below dams are often also altered from the expected natural state. Macroinvertebrate communities play an important role in stream ecosystem function and any impact on them may have implications for overall river health.

This observation has been attributed to altered temperature regimes and habitat below dams (Lessard and Hayes 2003) and adverse changes in water quality (Mantel et al 2010b). Temperature has important implications for feeding and metabolism, while dams act as sinks for fine sediment and woody debris and alter habitat through changes in flow regime. Temperature increases below dams are a particular problem in summer when increased summer temperatures may result in significant distances before temperature equilibrium occurs (Lessard and Hayes 2003). Hayes at el. (2006) found significant impacts on fish composition below dams where temperature increases above 2°C occur and the impact of temperature change below dams is considered at least as significant an impact as habitat fragmentation and barrier effects. A study of the impact of small dams in a Wisconsin watershed found that impacts of temperature and flow alteration are more significant in terms of impact on species richness than loss of connectivity (Cumming 2004). Kashiwagi and Miranda (2009) attributed fish composition change below small dams largely to habitat alteration.

Farm dams are known to alter downstream geomorphology of streams through altered flow regimes and the trapping of sediment, resulting in alteration of the physical habitat that supports instream biota (Davies 2003, Ligon et al. 1995). The capture of fine sediment can alter instream habitat while the reduction of flow or alteration of flow peaks and bed scouring flows can lead result in vegetation encroachment and changes to channel morphology (Mantel et al. 2010a, Power et al. 1996). The trapping of sediment can also lead to channel incision. Physical habitat is intimately linked with biodiversity and any change to the physical features of a stream is likely to have implications for instream and riparian ecosystems (Power et al. 1996). Farm dams can, however, have an attenuating effect on the increased sediment loads resulting from disturbance within catchments. Verstraeten and Prosser (2008) found an increase in sediment load within the Murrumbidgee Basin post European settlement of 370%, however farm dam storage reduced this to 250% and a similar effect was found in a study of farm dams in South Africa (Boardman et al. 2009). It should be noted this sediment load is not lost to the system and sediment is often removed from farm dams. Boardman et al. (2009) found that where sediment capacities in neglected dams reached capacity the sediment continued to be entrained downstream, In addition farm dam development may be associated with agricultural activities likely to lead to an increase in sediment loads.

3.1.2.2 Forestry

Catchment Yield

The impact of forestry on water quantity is well understood in general terms. A distinction needs to be made between afforestation, the establishment of plantations on grasslands or pasture, and native forest silviculture or regeneration, where the pre-existing vegetation type is native forest. If sufficiently extensive, the most significant impacts on a catchment scale are reduced yields and groundwater recharge and changes in seasonal runoff distribution, timing and magnitude of peak floes and persistence of low flows (Vertessy 2000).

The mechanisms and processes behind the impact of various vegetation types on runoff are well understood (Zhang et al. 2001). The impact on runoff is related to the extent of cover and the type of cover, for example, pine forests have a greater impact on runoff than native eucalypt plantations (Vertessy 2000). Afforestation reduces mean annual runoff through increased evapotranspiration and to a lesser extent through reduced groundwater recharge (Zhang et al. 2007). Afforestation has its greatest impact on absolute runoff with increasing annual rainfall and also for wetter than average years, however the greatest proportional reduction and greatest impact is in low-rainfall areas or at times of low flow (Vertessy et al. 2002, Duggan et al. 2008). The number of low or zero flow days is likely to increase following afforestation. Complete afforestation of small headwater catchments with mean annual rainfall of around 900 mm potentially increases the number of zero-flow days from a range of 0-50 to a range of 175-225 days per year (Zhang et al. 2007, 49). Where annual rainfall is 1 500 mm an equivalent 2 ML more water per year per hectare of forest is lost from runoff compared with grasslands (Keenan et al. 2004). The difference is considered undetectable where the annual rainfall is less than 500 mm. Vertessy et al. (2002, 105) provide some basic calculations that demonstrate that at the 1000 mm isohyets a 100 ha pine plantation would reduce mean annual runoff by about 300 ML, or the equivalent of 100 small farm dams. This assumes an average farm dam volume of 1.2 ML and an average flow reduction of 2.5 ML per ML of dam storage. In an 800 mm rainfall zone, conversion from annual pastures to trees results in an average water yield reduction of about 1.5 ML for each hectare planted (van Dijk et al. 2006).

The effect for a particular location will depend on soil type, topography and position of the forest area in the landscape. The impact of afforestation on runoff varies with tree age, with runoff reductions minor for the first five years after afforestation and greatest 10-20 years after planting (Keenan et al. 2004). Water yields from forests then slowly increase after 30 years of age with the decline in association with reduced growth rate. Fast growing plantation species in particular have often been found to cause significant reductions in catchment flows (Calder 2007). In addition, while impacts at a regional or whole of catchment scale may be of a lower order of magnitude, impacts at a sub-catchment spatial scale where there is a significant plantation area may be significant, both in terms of end of system flows and allocated water (Brown et al. 2007). In addition this impact is likely to be exacerbated during periods of low flows and in critical years.

In native forest regeneration harvest systems, harvesting generally results in an increase in water yield followed by a decline as the mechanisms detailed above begin to dominate as the regenerating forest or plantation begins to grow (Barmuta 2006). Water yields will return to the pre-harvest condition as the forest ages. This relationship is illustrated in the following Figure 3.1 from Brown et al. (2006). Barmuta (2006) provides five caveats on this general result; it applies to the harvesting system of clearfell-burn-and-sow (CBS), different forest management may impact evapotranspiration, it relates to regenerating forest replacing mature forest, it will vary according to catchment conditions, stochastic events such as bushfires and to climate change. The qualitative changes to flow yield and pattern for broad categories of forest management in Tasmania is given in Table 3.1, taken from Barmuta (2006).

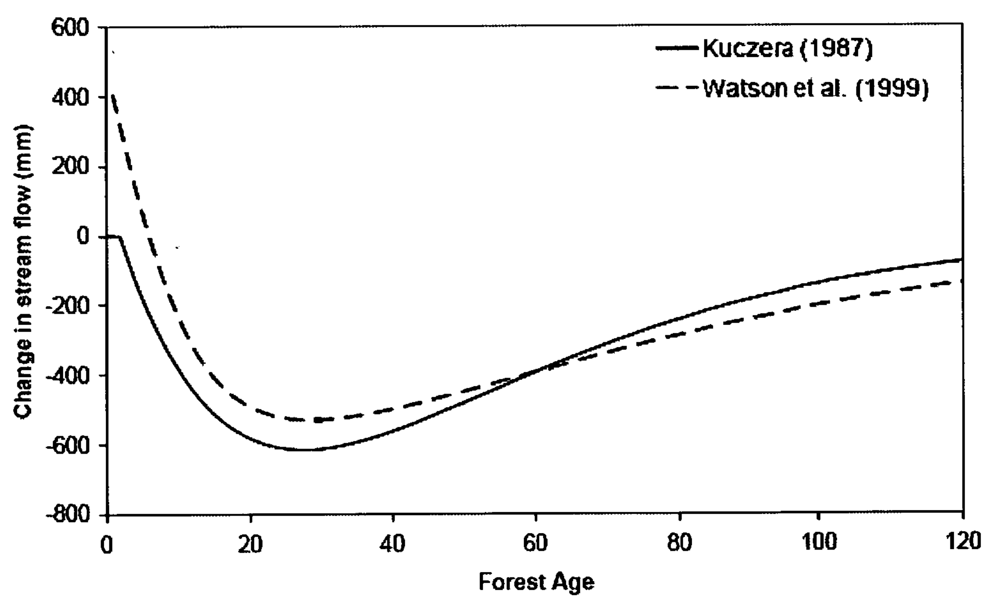


Figure 3.1 Change in streamflow relative to pre-harvest conditions as a function of forest age (for *Eucalyptus regnans* receiving 2000 mm/y precipitation). Source: Brown et al. 2006, cited in Barmuta 2006.

Table 3.1 Qualitative changes to flow yield and pattern for broad categories of forest management in Tasmania. Source: Barmuta (2006).

Type of forest management	Changes to water yield relative to pre-forestry land use	Changes to flow patterns	Potential problems for aquatic biodiversity	Potential benefits for aquatic biodiversity
Native forest harvesting and regeneration	Timing and magnitude of the effects depend on the species and prevailing climate. Patterns could be much attenuated in shelterwood and selective cut systems.	Depends on age of regenerating forest. Initial intensification of high flow events immediately after harvest, shifting towards drier flow duration scenarios during most intensive phase of regeneration. Longer-term responses depend on forest management.	Habitat changes from initial increases in flow after harvest: inputs of fine sediment, increased competence of stream may increase channelization and change proportional representation of in-stream habitat types. Potential for increased frequency and duration of low flows and cessation of flow during years of intense water use by regenerating forest.	Some indirect benefits (e.g. off-stream fire dams may provide drought refuge for amphibians).
Conversion of native forest to plantation	Depends on species involved. Pine plantations probably lose more water to evapotranspiration than native eucalypts, but there are insufficient data for eucalypt plantation species in Australia.	Depends on species involved. Pine plantations may shift flow duration curve to drier pattern than native eucalypt forest. Little empirical data available applicable to Tasmania.	In-stream habitat changes from increased runoff during establishment phase. Loss of habitat if plantation species or management increases frequency or intensity of low flow events during dry season.	
Conversion of pasture or cropped agricultural land to plantation	Decrease in yield relative to agricultural conditions	Depends on age of plantation and rotation times. General shift to lower flows that intensifies as	Dewatering of smaller streams during dry spells and seasons. Cumulative effect downstream	Return to a pattern of flows more similar to that prior to agricultural use. Restoration of

Type of forest management	Changes to water yield relative to pre-forestry land use	Changes to flow patterns	Potential problems for aquatic biodiversity	Potential benefits for aquatic biodiversity
		water requirements of trees increase. Low flows tend to be more frequent and more prolonged; high flows less intense. Longer term-responses poorly documented with empirical data relevant to Tasmania.	could compromise longitudinal connectivity (e.g. fish passage) in river segments remote from forestry activity.	native riparian species to stream sides.

Biodiversity

Changes to the hydrological characteristics of streams are likely to result in impacts on dependent biota through habitat alteration, water quality changes and life cycle triggers. However, the impacts of forestry on in-stream biota as a result of changes to flow regime are difficult to unconfound from other land use impacts that may also have a cumulative effect across the catchment (Barmuta 2006). In particular the cumulative downstream effects of forestry on reaches remote from forestry operations remains a knowledge gap as does the biota of ephemeral and temporary streams in forested landscapes in Tasmania.

On area of impact that is better documented is the effect of sediment inputs. Increased sediment fills interstitial spaces which can alter benthic community composition and may also effect the recruitment and growth of some fish and amphibians (Barmuta 2006). Sediment inputs are not uniform across catchments; they vary with rainfall and are also related to delivery pathways such as minor gullies that allow for direct connectivity. The spatial distribution and relative contribution of sediment from both dispersive and direct pathways and their relationship to features of forestry operations such as road drains can be explicitly modeled and quantified (Croke at el. 2005). In some catchments the road network associated with forestry operations delivers more sediment than harvest areas, with one study determining that one road section (4m x 100m) produced as much sediment as a 30 ha logged catchment (Sheridan and Noske 2007). However, a study of a forested catchment in south east Australia found the impact of forest roads on sediment and nutrient input to be minimal (Sheridan and Noske 2007). Modeling of sediment inputs to the Tamar Estuary in northern Tasmania attributed the majority of the total suspended solid load to forestry and grazing, with forestry contributing over twice that of grazing (BMT WRN 2010). There was a disproportionate contribution from forestry operations in steeper terrain with higher rainfall. Sediment input into streams can also be considered as a proxy for nutrient and pesticide inputs.

Riparian zones provide most of the energy base for aquatic systems in forested catchments and are important in the regulation of light and temperature. Tasmanian research indicates statistically defensible changes in macroinvertebrate community structure tend to occur where riparian buffers are less than 30 m (Barmuta 2006). Under the current Tasmanian Forest Practices Code (the code), buffers for streams defined as class three under the code are provided with streamside buffers of 20m (Forest Practices Board 2000). In small headwater streams, generally defined as class four streams, a 10 m machinery exclusion zone is provided. While this is likely to be beneficial, particularly in mitigating sediment delivery, it is considered that there are clear impacts on ecosystem processes in headwater streams (Barmuta 2006). Buffers are vulnerable to wind throw, fire from regeneration burns and sediment inputs from roads and snig tracks.

In Tasmania, the macroinvertebrate biota of headwater streams generally consist of a subset of that which occurs downstream with a general absence of aquatic vertebrate fauna although in favourable conditions headwater streams may represent important components of the range of native fish and may also support platypus. There can be a high degree of local endemism in freshwater or freshwater dependent or invertebrates. Barmuta (2006) suggests that low biodiversity in Tasmanian headwater streams cannot be assumed, with potential for regional patterns of endemism.

Forestry activities have measurable impacts on the ecology of Tasmanian headwater streams. Davies et al. (2005) observed substantial differences in benthic macroinvertebrate community composition and abundance, aquatic insect emergence rates and macrophyte and algae abundance between logged streams and control streams where the logging had occurred 15 years previous to the study. Differences in riparian vegetation structure, channel form and sediment composition were also apparent. It should be noted that the logging occurred prior to the relevant prescriptions in the code. More recently, Clapcott and Barmuta (2010), through a study of changes to metabolism and organic matter processes in forested headwater streams in Tasmania, found that current management practices do not protect instream processes from forestry processes within a two to five year time frame. The study suggested that forestry has a significant effect on metabolism and organic matter processing in small headwater streams. The cumulative impact of this effect may alter downstream processes through a catchment. A Tasmanian study of stream macroinvertebrates across paired forested streams found substantial differences in macroinvertebrate community composition and abundance (Smith et al. 2009). The changes were indicative of a shift from a depositional stream environment to a higher power erosional environment. A similar study of coastal streams in British Columbia, Canada, found stream morphology and community composition and abundance, with measurable impacts evident up to 40 years after logging (Zhang et al. 2009). A summary of the effects of forestry on headwater streams, from Barmuta (2006, 48), is given in Table 3.2

Table 3.2 Summary of the response of headwater streams to forestry activities. Source: Barmuta 2006.

Feature	Changes attributed to harvesting	Responses
Low hydraulic power	Higher peak discharges; flashier flows.	Increased scour of previously benign habitats. Probably more increased exports of organic matter and nutrients, longer nutrient spiralling paths.
Geomorphology	Increased incision or channelisation of stream bed. In extreme cases, increased erosion of bed and banks; bank failure; increased probability of debris flows.	Change of proportional representation of habitats. In extreme cases, dramatic alteration of in-stream habitat.
Flow paths and temperature	Routing of overland flows into streams via roads, drainage lines	Sedimentation; interstices filled with fine (inorganic) sediment. Altered microbial functioning.
Flow seasonality	Peak winter high flows may be higher; unclear whether, low flows or dry periods are prolonged.	Elevated delivery of sediment during peak discharges
Disturbance regimes	Vulnerable to low flows, landslides or debris flows	Habitat loss and isolation
Recolonization pathways	In-stream movements may be impeded by slash, fallen wood; riparian changes may alter flight patterns.	Recruitment loss or failure of fish species; reduced dispersal
Aquatic-terrestrial linkages	Reduced habitat for terrestrial fauna immediately after harvest; less allochthonous coarse particulate organic matter (CPOM). Regrowth/regeneration in riparian zone may favour different mix of plant species	Energy flows to terrestrial compartment may be reduced; possible reduction of secondary production delivered to downstream reaches. Changed nutrient regimes if recolonising spp. fix nitrogen
Aquatic vertebrates	Temperature effects on spawning or recruitment of fish & amphibians; effects may carry downstream. Changes to secondary production; sometimes short-term increase immediately after harvest followed by declines later.	Decreased fish or amphibian abundance. Platypus absent from previously harvested reaches.
Canopy closure	Light reaching stream bed increases initially followed by decline as canopy closes over. Initial decrease of leafy CPOM, but accidental inputs of large woody debris (LWD) may result	Initially, localised benthic algal blooms with increased use of these resources in food webs. Initial decline in retention of leafy CPOM or an increase if LWD increases retentiveness. Less

	from harvesting or increased wind throw during regeneration and regrowth.	CPOM for a period prior to re-establishment of larger woody riparian species

3.1.2.3 Water Extraction

The extraction of water for consumption purposes and collection acts to 'dampen' the magnitude, frequency and duration of smaller flow events and low flows and therefore, fundamentally, results in the alteration of natural flow regimes (DPIW 2007a). In this broad hydrological sense the impacts of water extraction on the river systems are of the same nature as that of farm dams and forestry. The following discussion then can be equally applied to these and any other process that alters natural flows. The role of river flow in governing ecological processes and impact of anthropogenic activities on natural flow has been the subject of extensive research. It is nonetheless an area where understanding continues to be challenged due to the complexity of the many interactions within a catchment between the flow regime and flow dependent ecosystems and species. It is not the intention here to provide a comprehensive review of this matter but rather, for completeness, to provide a broad overview in order to place the impact of water extraction within the context of cumulative impacts on the natural flow regime.

Stream flow has been described as a 'master variable' (Poff et al. 1997) as it limits the abundance and distribution of riverine species. It is strongly co-related to chemical and physical characteristics of rivers such as temperature, geomorphology and physical habitat and diversity. Characteristics of a river's flow are evident in water quantity, timing and variability and these factors may be evident across a considerable temporal scale. Five critical components are considered to regulate ecological processes: the magnitude, frequency, duration, timing, and rate of change of hydrologic conditions (Poff et al. 1997). Flow variability is considered as the key to maintaining geomorphological and biological components of riverine ecosystems, with the natural flow regime of a river representing the optimum variability for that river's ecosystem DPIW (2007a). Ward (1989) conceptualises this high level of spatial and temporal heterogeneity in terms of an interactive pathway system along four dimensions. Connectivity occurs longitudinally, laterally (channel and riparian zone/flood plains), vertically (channel and groundwater), with the time the fourth dimension. The influence of flow regime on aquatic biodiversity is presented by Bunn and Arthington (2002) in terms of four overarching principles:

- flow is the major determinate of physical habitat and therefore biotic composition at various spatial scales;
- aquatic species have evolved life history strategies in response to the natural flow regime;
- maintenance of the natural patterns of connectivity are essential to ensure the viability of populations of many riverine species; and
- invasion and success of exotic species is facilitated by altered flow regimes.

Change to the natural flow regime therefore has the potential to impact on any physical or biotic component of the riverine landscape and associated ecosystem processes. It can alter in-stream habitat at various spatial and temporal scales for invertebrates, fish and macrophytes (Bunn and Arthington 2002, Poff et al. 1997). Invertebrate richness commonly decreases in response to decreases in habitat diversity with reduced flows also leading to increased drift in response to stress (Dewson et al. 2007, Matthei et al. 2010). Impacts on fish in particular and other aquatic biota can be related to life cycle triggers or requirements either because of altered habitat or due to the change in flow. The impact in life history patterns can also be observed for aquatic plants, while the variability of the lateral flow connectivity has important implications for riparian species and altered flow regimes may result in changes in riparian community composition (Bunn and Arlington 2002). Alteration to the flow regime will also result in changes to the relationship between geomorphology, which is driven by flow, and the riparian and floodplain zone. These features in turn may also influence the pattern of flow and associated deposition and erosion (DPIWa 2007). This relationship is summarised in Table 3.3. Water abstraction is a single stressor that is likely to be acting in the presence of other stressors. Matthaei et al. (2010) found that sediment, nutrient and water abstraction stressors interact often, particularly sediment and flow, with the impact of these stressors most evident at low flow. Water abstraction in streams subject to high fine sediment was found to have a far worse impact on invertebrate fauna than streams with low sediment levels.

Table 3.3 Hydrological events, corresponding geomorphological and biological features and anthropogenic impacts. Source: (DPIW 2007a, 10).

Temporal scale	Responding geomorphological feature	Responding biological feature	Anthropogenic impacts on natural flow regime
> 100 years	Catchment	Biotic ecosystem	
1-100 years	Floodplains, valleys, channel morphology	Riparian and floodplain communities	Large instream dams
< 1 year	Instream and bankside morphology	Instream and bankside communities	Instream dams, flood abstraction, farm dams, irrigation abstraction
< 1 day	Substrate structure	Benthic and hyporheic communities	Irrigation abstraction, stock and domestic abstraction

3.2 Catchment Case Study: Great Forester - Brid

3.2.1 Introduction

The Great Forester–Brid catchment in north eastern Tasmania is 772 km^2 (Figure 3.1) with the Great Forester catchment area comprising 520 km^2 (CSIRO 2009a). The climate of the north east is heavily influenced by the maritime environment with rainfall patterns varying with topography. Mean annual rainfall is 982mm, and near the coast, annual average rainfall is about 750 - 800 mm increasing to around 1200 mm in the upper catchment (Graham 1999a). Highest monthly rainfall occurs in July and August with the lowest in February and March.

The Great Forester River rises in elevated, steeply forested country in excess of 1000m elevation and passes through hilly, forested and agricultural areas with floodplain development in the middle catchment before entering extensive lowland plains with swamps and sand dunes. The Brid River originates from mid elevation (Mt Scott at 660 m) and quickly descends to lower elevation agricultural land that has been cleared for a mixture of cropping and pasture (DPIW 2009a). The middle catchment is heavily forested with the lower catchment largely cleared for pasture (Graham 1999b). The Great Forester - Brid catchment is used extensively for forestry (native forest harvesting and plantations) and agriculture (dairy, beef and sheep grazing as well as intensive cropping and hop production) with only minor industrial uses such as timber milling (McKenny and Read 1999). Population is low, with the major town, Scottsdale, having a population of approximately 2000. Land use is given in Figure 3.2.

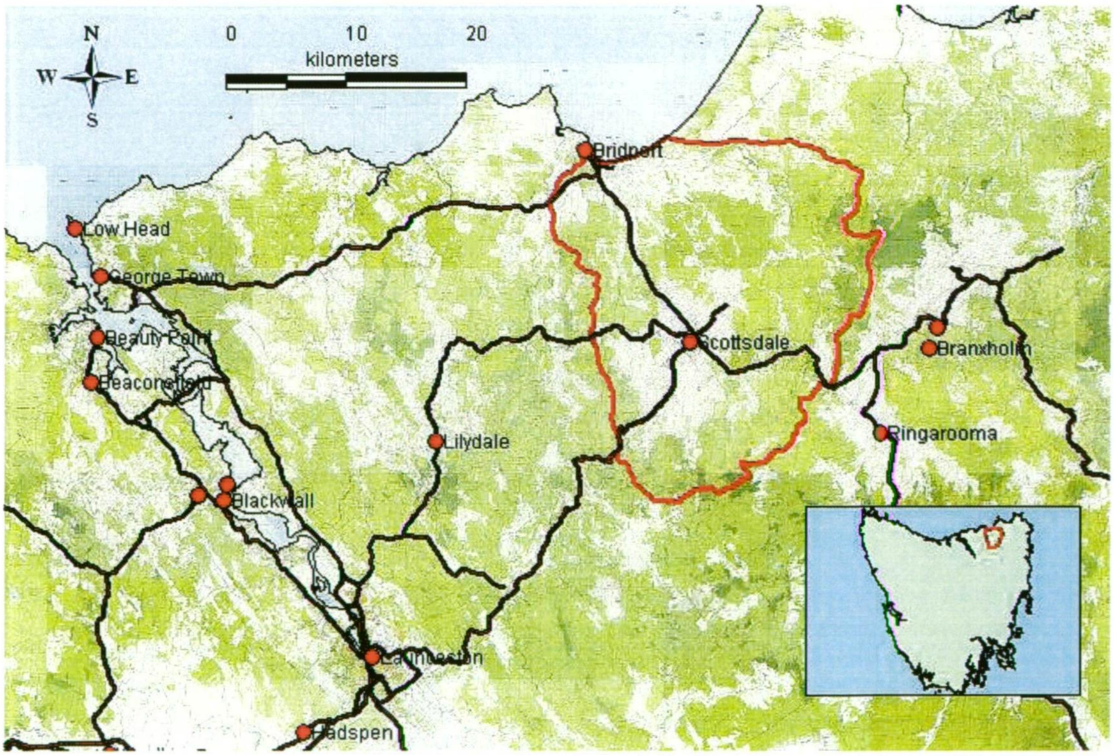


Figure 3.1 Location of Great Forester-Brid catchment.

Mean annual flow for the Great Forester River at the stream gauge in the middle catchment (Forester Road bridge, gauge #19201) between 1971 and 2003 was 220 ML/day, between 2004 and 2007 it was 154 ML/day, with a maximum of 407

ML/day (1975) and a minimum of 93 ML/day (2006) (DPIW 2007b). Mean monthly flows range from 67 ML/day in February to 424 ML/day in August. The upper catchment stream gauge (Prosperity Road, gauge #19224) was installed in 2002 and has recorded annual mean flows of between 37.2 ML/day and 57.7 ML/day (DPIW 2007b). Average daily flows for the Brid River recorded at the stream gauge above the tidal limit (Brid River 2km upstream tidal limit, gauge #19200) over the previous ten years is 105 ML/day. The locations of the gauging stations are shown in Figure 3.3.

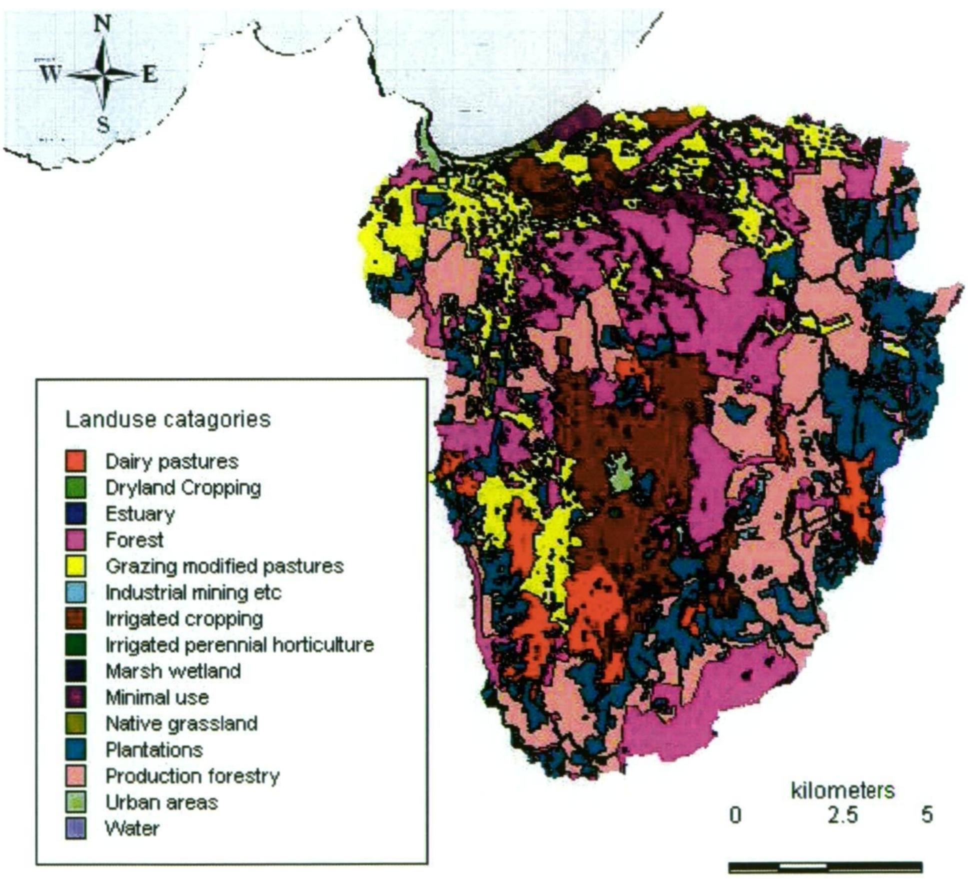


Figure 3.2 Land use in the Great Forester – Brid catchment.

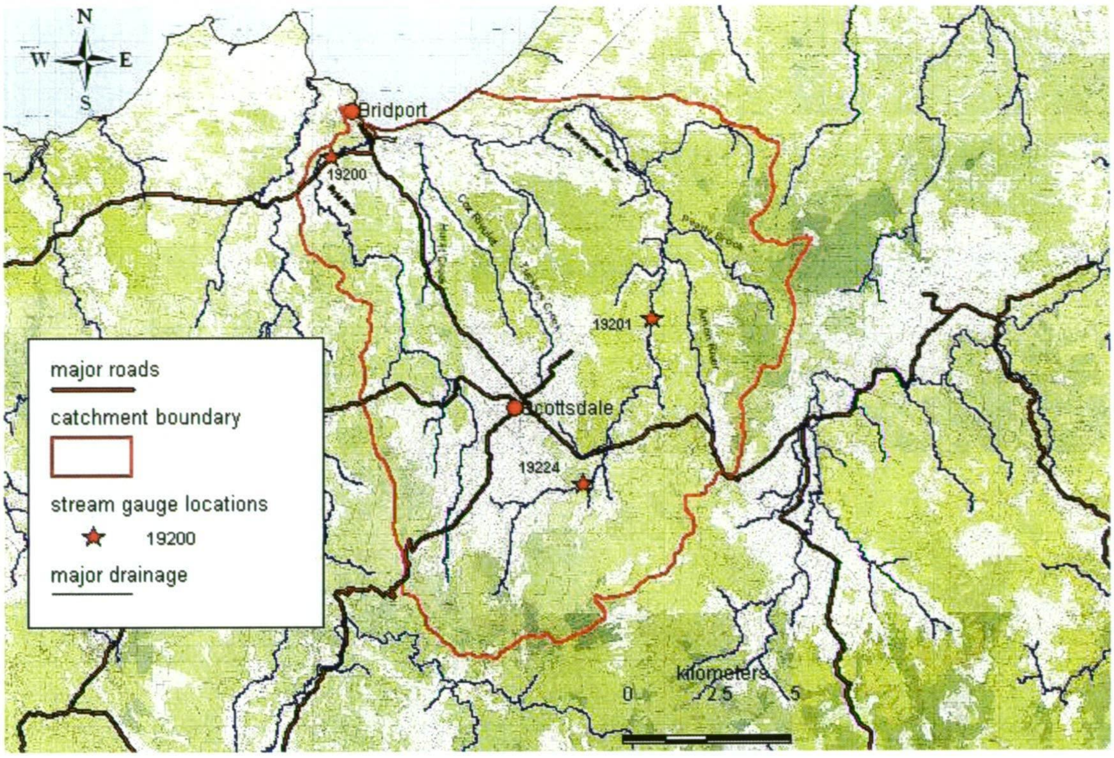


Figure 3.3 Major drainage Great Forester – Brid catchment.

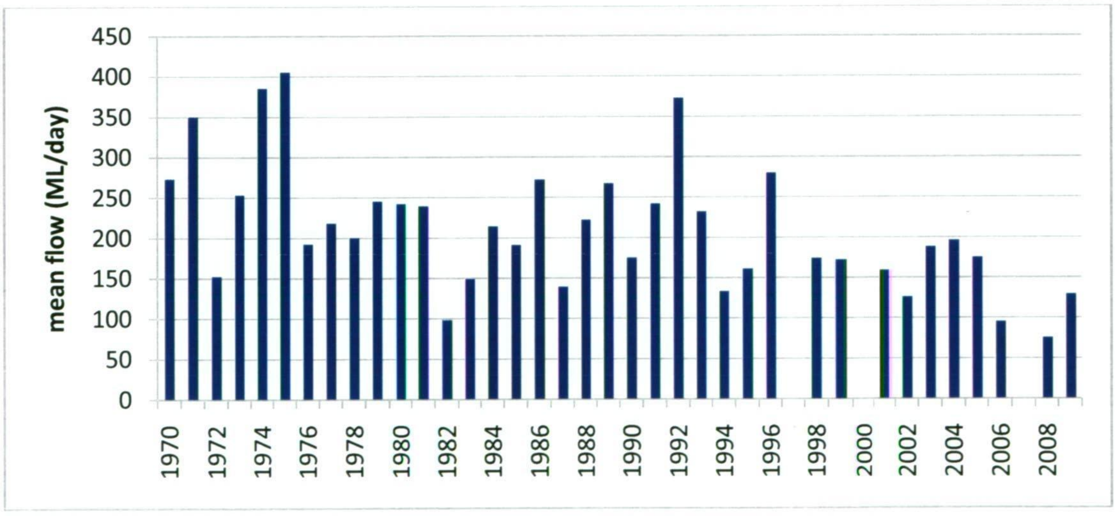


Figure 3.4 Mean flow Great Forester River stream gauges.

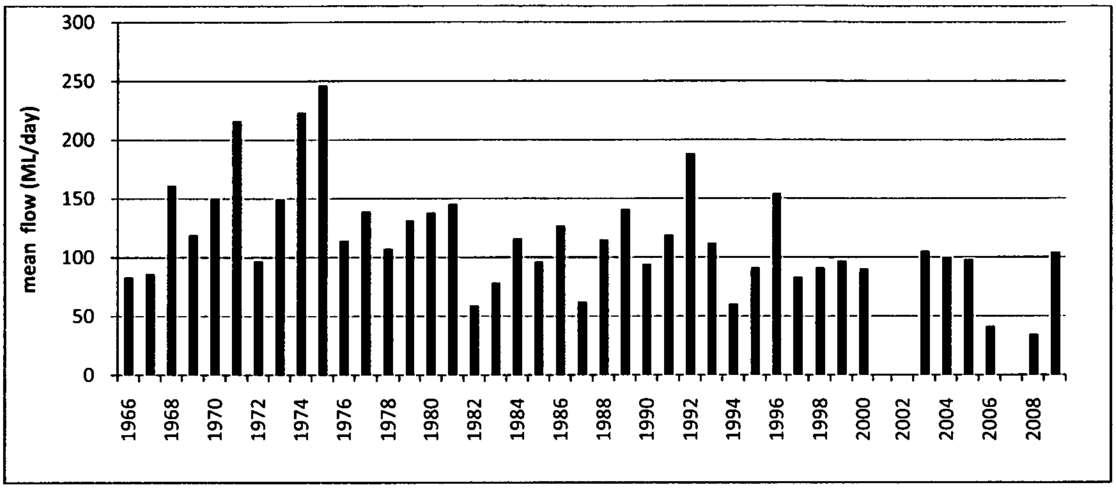


Figure 3.5 Mean Flow Brid River stream gauge.

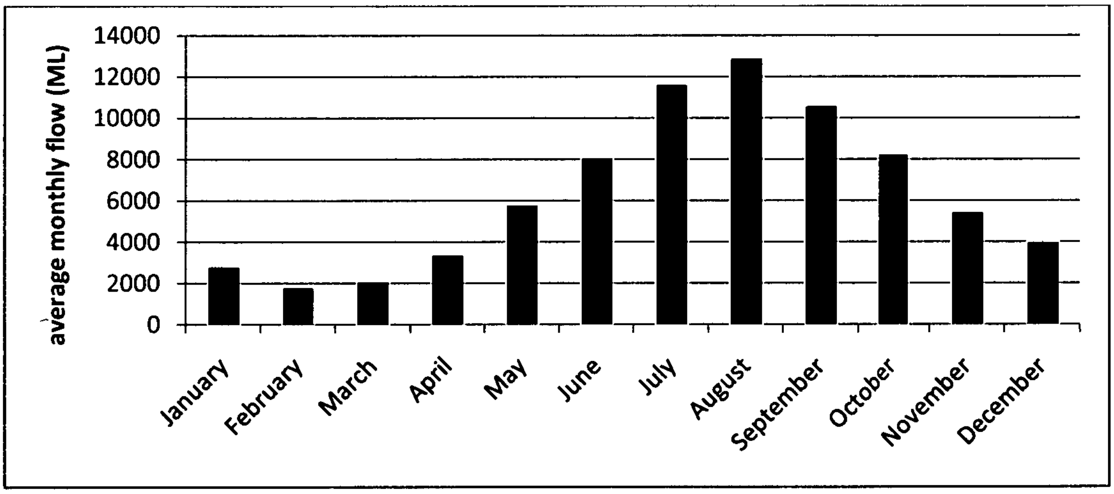


Figure 3.6 Average monthly flows at the Great Forester River stream gauge 19201.

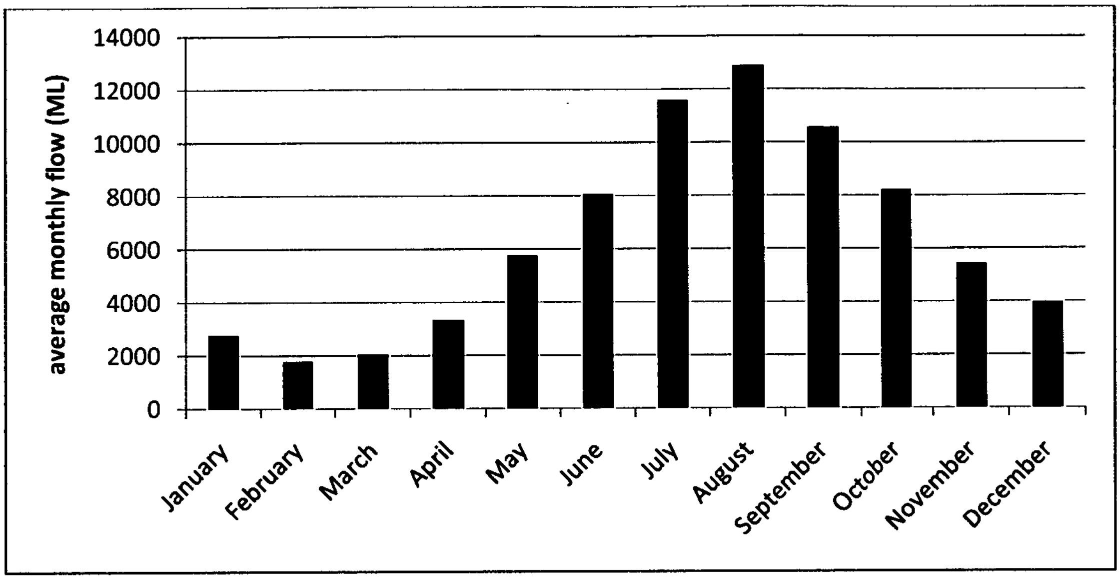


Figure 3.7 Average monthly flows at the Brid River stream gauge 19200.

3.2.2 Farm Dams

3.2.2.1 Number and Distribution

Farm dam data utilised in this section was derived from the Water Information System Tasmania website (DPIPWE 2010a). In total there are 264 licensed dams known to exist within the catchment with a total capacity of 5666.47 ML (Table 3.4). A further 66 dams have been approved. Previously there was not a process in place to enable the construction of dams to be verified. However, the construction of recent dams can be verified through the submission of a report upon completion as a permit requirement. Dams approved prior to this process are considered to exist by the Department of Primary Industries, Parks, Water and Environment (DPIPWE), while recently approved dams are also considered in the analysis on the assumption that they are likely to exist in the near future. In total there are 330 approved dams with a total capacity of 11105.6 ML with a dam density of 43 dams per 100 km^2.

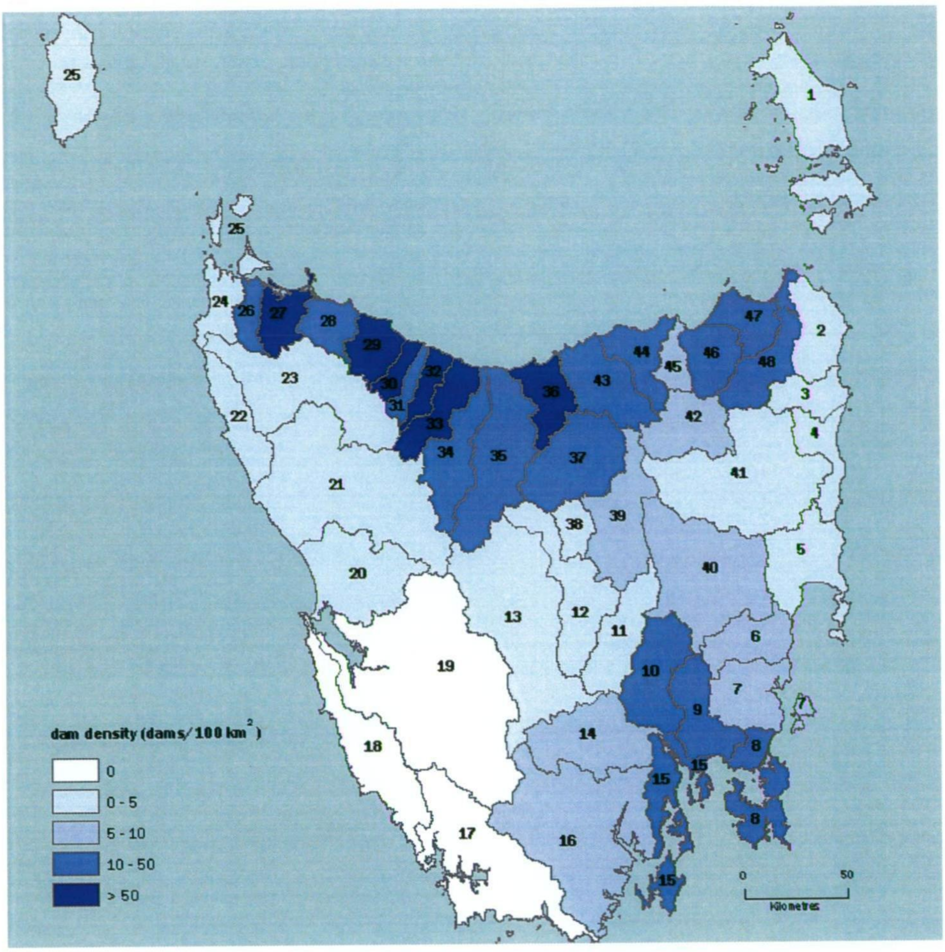
Table 3.4 Farm dam types and capacity for the Great Forester-Brid catchment.

Type	No. of Dams	Capacity (ML)
Irrigation	237	5532.52
Stock and Domestic	24	119.7
Other	3	14.25
Total	264	5666.47
Proposed	66	5439.15
Total	330	11105.62

From a State wide perspective, the dam density of the Great Forester – Brid catchment is high (Tasmanian Planning Commission 2009). In the northeast of the State the adjacent catchment of Ringarooma has the next highest density at 15.9 dams per km^2. The dam density of the Great Forester – Brid catchment is exceeded only by the intensive agricultural catchments of north western Tasmania. Dam densities of selected catchments are shown in Table 3.5. The distributions of dams across Tasmania in terms of dam density are shown in Figure 3.3. These figures exclude Hydro Tasmania dams.

Table 3.5 Catchment dam densities across Tasmania

Catchment	Catchment Area	Total Dams	Density (dams/100km^2)
George	634.2	14	2.2
Clyde	1119.3	50	4.5
Macquarie	2728.9	165	6.0
North Esk	1063.7	83	7.8
Pipers	751.6	90	12.0
Ringarooma	10006.7	160	15.9
Black – Detention	584.6	259	44.3
Rubicon	731.7	468	64.0
Leven	773.9	669	86.4
Inglis	617.0	603	97.7
Blythe	373.2	385	103.2



1 Furneaux	11 Clyde	21 Pieman	30 Cam	40 Macquarie
2 Musselro – Ansons	12 Ouse	22 Nelson Bay	31 Emu	41 South Esk
3 George	13 Upper Derwent	23 Arthur	32 Blythe	42 North Esk
4 Scamander – Douglas	14 Lower Derwent	24 Welcome	33 Leven	43 Tamar Estuary
5 Swan – Apsley	15 Derwent Estuary - Bruny	25 King Island	34 Forth – Wilmot	44 Pipers
6 Little Swanport	16 Huon	26 Montagu	35 Mersey	45 Little Forester
7 Prosser	17 Port Davey	27 Duck	36 Rubicon	46 Great Forester – Brid
8 Tasman	18 Wandered – Giblin	28 Black – Detention	38 Great Lake	47 Boobyalla - Tomahawkd
9 Pitt Water – Coal	19 Gordon – Franklin	29 Inglis	39 Brumbys – Lake	48 Ringarooma
10 Jordan	20 King – Henty			

Figure 3.8 Distribution of non Hydro dams in Tasmania as dam density. Source: Tasmanian Planning Commission 2009.

The 292 dams constructed for irrigation purposes comprise 8924.7 ML or 80% of total farm dam capacity. Eleven dams are catchment dams, the remainder are instream dams. The frequency distribution based on capacity is shown in Figure 3.9 239 dams below 20ML contribute 15% of total capacity, 75 dams between 20 and 100 ML capacity contribute 23% of capacity, while the 14 dams above 100 ML contribute 53%.

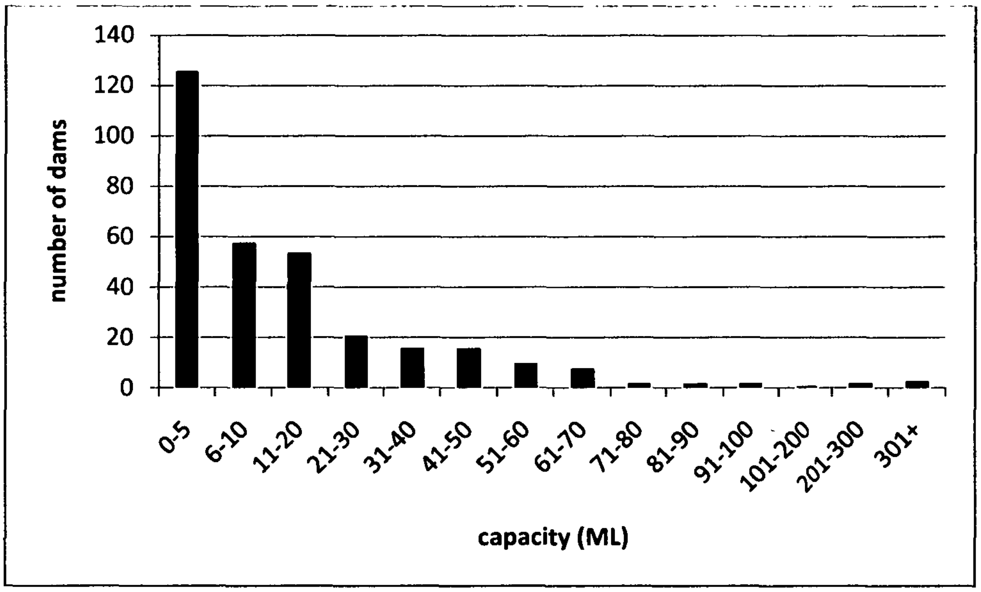


Figure 3.9 Frequency distribution of dams in the Great Forester-Brid catchment.

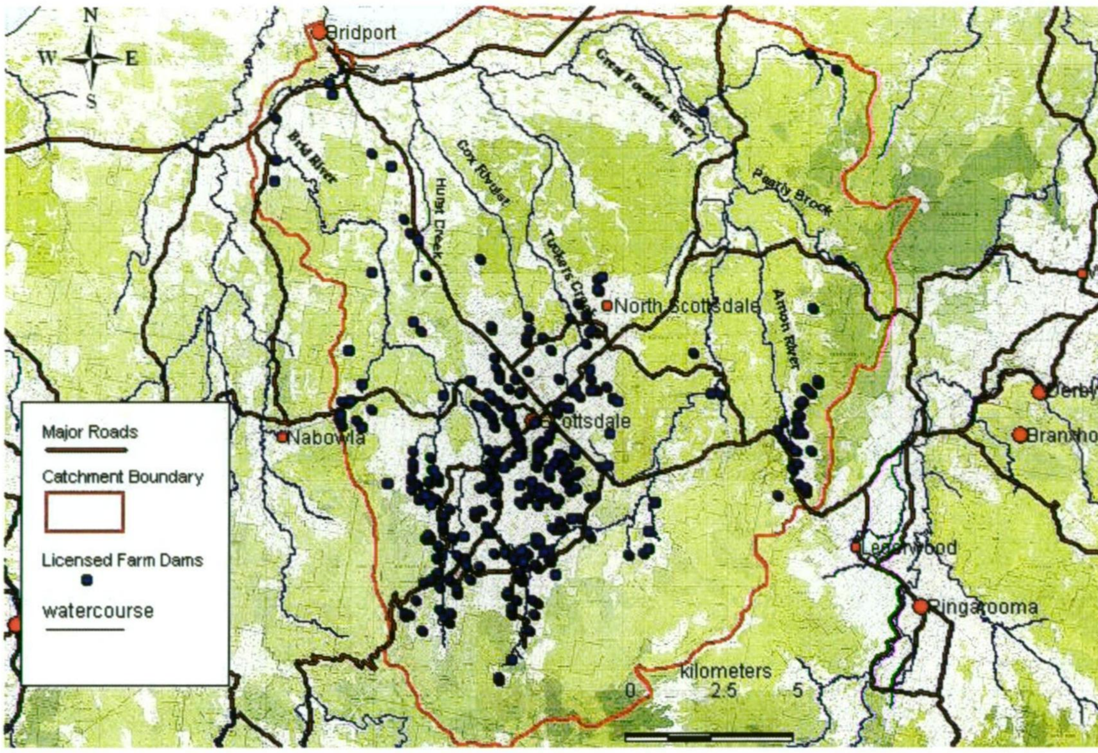


Figure 3.10 Farm dam locations Great Forester-Brid catchment.

The *Water Management Act 1999* is the legislative mechanism for dam construction in Tasmania. A permit is not required for stock and domestic dams that are under 1 ML (prior to 2000 it was 2.5 ML) in capacity that are not on a watercourse (as defined in the Act). In addition it is also probable that there are unlicensed illegal dams present within a given catchment. An estimate for the number of farm dams that fall into these categories in the Great Forester catchment has been undertaken by Hydro Tasmania Consulting (2008a). This work utilised Google Earth images, mostly dated from 2005, to determine dam locations by eye. A number of unlicensed dams were identified and it was assumed that the capacity of these dams could range up to 20 ML. In total 247 unlicensed dams were identified. Using work from Neal et al. (2002) it was assumed that the average capacity for dams less than 20 ML was 1.4 ML. With a usage assumed at 100%, the total volume was estimated at 345.8 ML. Dams were assumed to be empty in May and refill once over the winter.

A similar study for the Brid catchment found an additional 129 unlicensed dams (Hydro Tasmania Consulting 2008b). The same assumptions result in a capacity of 180.6 ML. The total unlicensed farm dam capacity derived from this work constitutes 4.7% of the current total licensed farm dam capacity. This work must be seen as an approximation only, as dams were identified from aerial photography and numbers extrapolated from the sub-catchment scale. In addition the work of Neal et al. (2002) may not transfer particularly well to the Great Forester – Brid catchment given the different catchment characteristics.

It is clear from Figure 3.10 that farm dams are not distributed evenly throughout the catchment. Three areas of high dam density (Figure 3.11) have been identified. The breakdown of farm dam numbers and capacity for each of these areas is given in Table 3.6. Figures 3.12 and 3.13 show these clusters in detail. The examination of these dam clusters is utilised in subsequent sections to consider impacts at the sub-catchment scale.

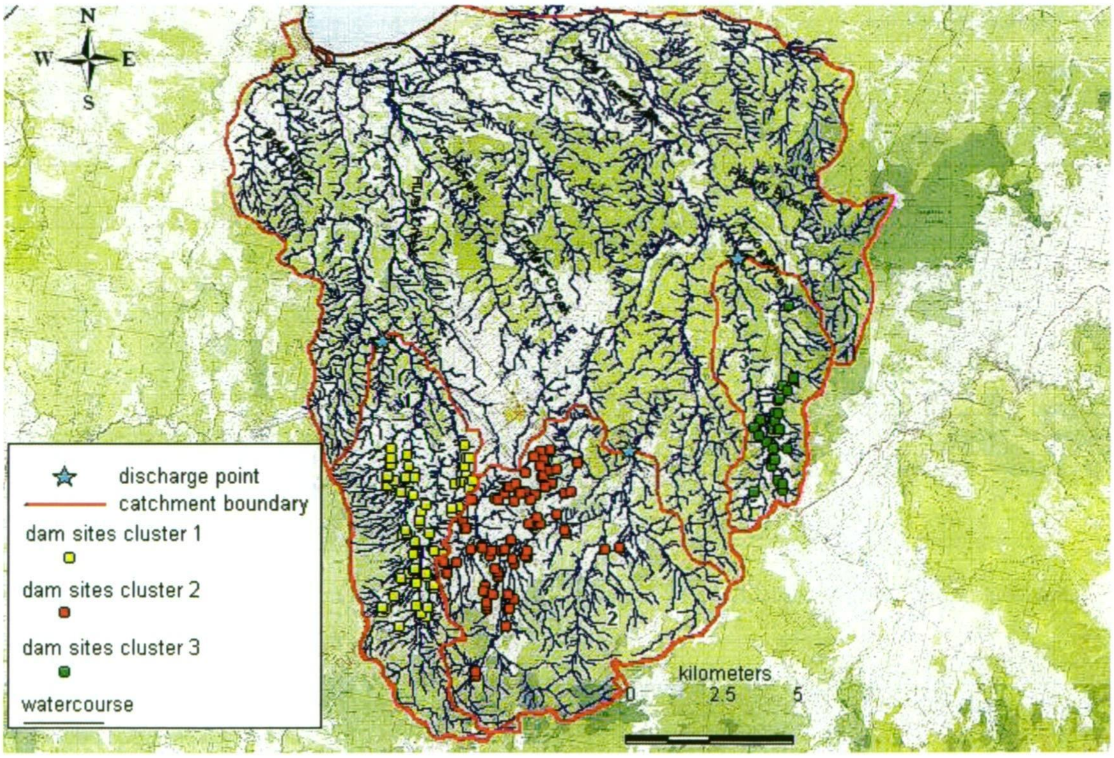


Figure 3.11 Dam cluster sub-catchments.

Table 3.6 Farm dam details for the three clusters shown in Figure 3.10.

Cluster	Number of Farm Dams	Capacity (ML)	Catchment Area km^2	Dams per 100 km^2
One (Brid)	73	1360	85	86
Two (Upper Great Forester)	93	4047	138	67
Three (Arnon River)	28	544	35	80

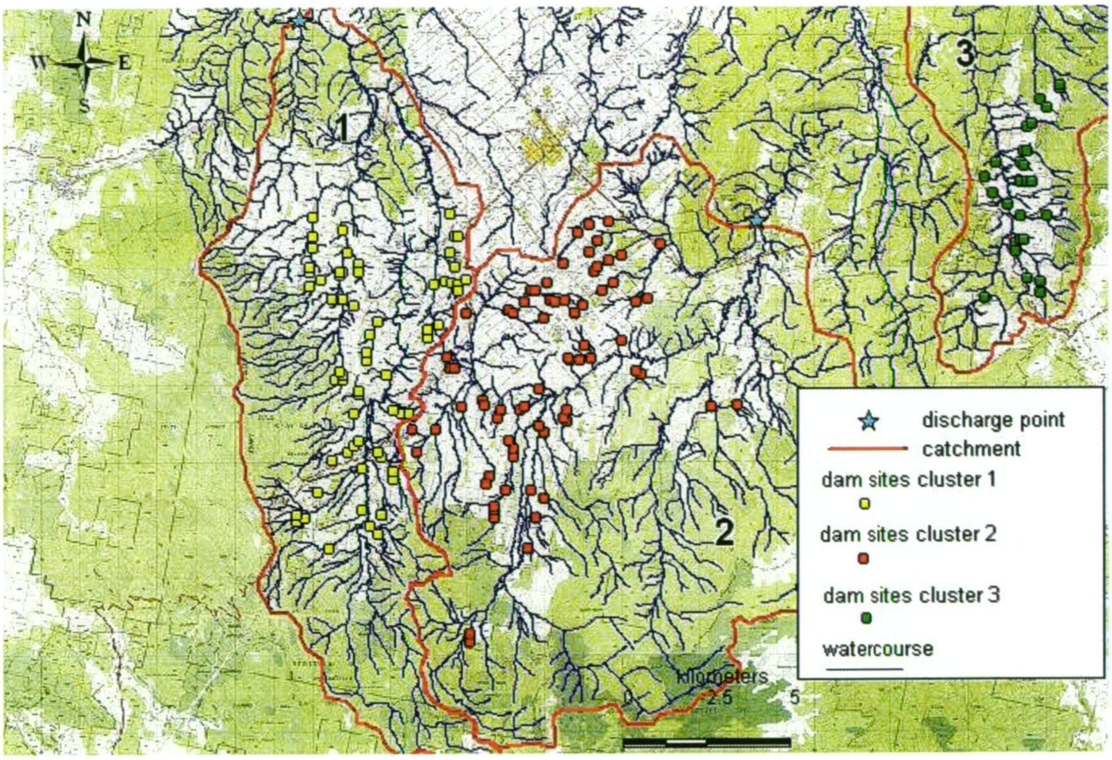


Figure 3.12 Cluster 1 and Cluster 2 farm dam sub-catchments.

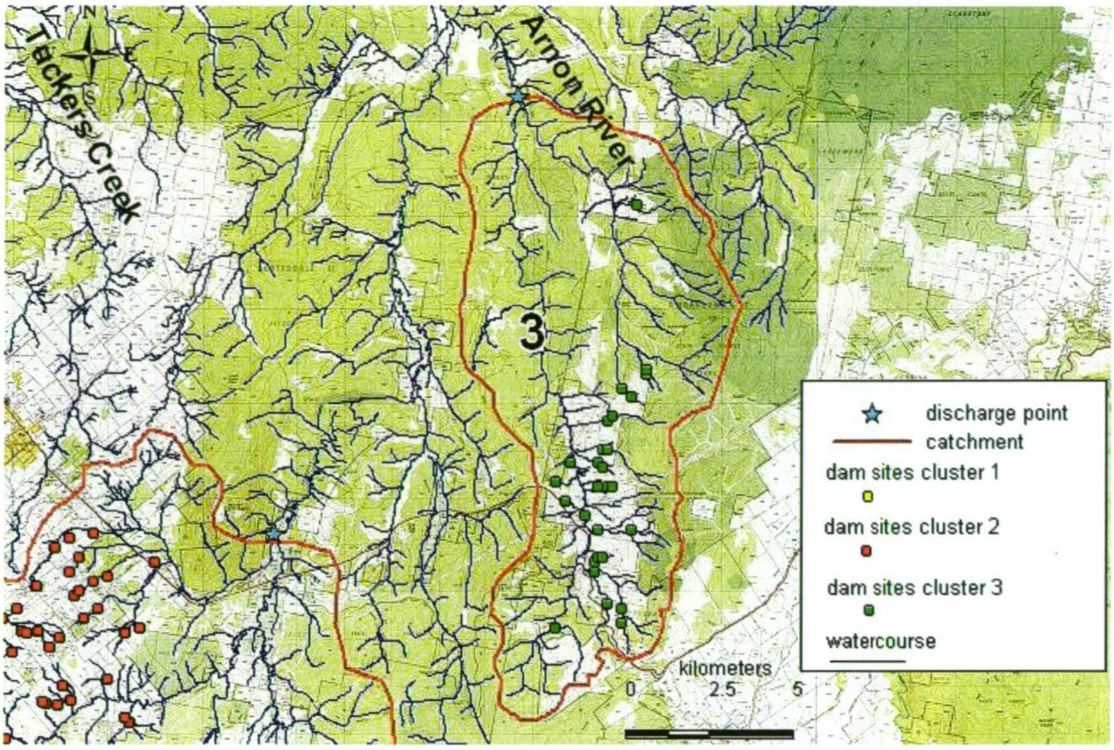


Figure 3.13 Cluster 3 farm dam sub-catchments.

A similar approach to that employed by Hydro Consulting (2008a and 2008b) was used to determine unlicensed dam numbers for Cluster 1. High quality aerial photography dating from 2009 was used in conjunction with a grid and licensed dam locations to estimate unlicensed dam numbers. As discussed above, there are a number of confounding features of aerial images such as tree clusters and shadows and this is an approximate only. This method identified 48 dams, many were small, however some were clearly greater than 1 ML. Plates 3.1 to 3.4 show some examples of unlicensed dams identified through this process.

Plate 3.1 Unlicensed dam, approximately 0.11 ha in area.

Plate 3.2 All the dams in this image (approximately 1 km^2) are unlicensed.

Plate 3.3 Unlicensed dam, approximately 2 ha in area.

Plate 3.4 The uppermost and middle dams are unlicensed. The dam on the lower right is a licensed 10 ML dam.

3.2.2.2 Catchment Yield

Without known usage data it is not possible to determine the absolute cumulative impact of farm dams on yield. Assuming a usage rate of 100%, the total losses from the system from licensed farm dams comprises 11105.6 ML. This represents 4.3% of the total annual catchment outflow of 256 000 ML. Water licenses for farm dams generally limit the taking of water for storage to the period May to November, inclusive. Licensed farm dam capacity is 7.4% of the total outflow for this period. Licensed dam capacity as a percentage of the catchment outflow for the period May to June is 20.2% and 47% of the total catchment flow for May.

It can be expected that at the spatial scale of the clusters of higher dam density identified in Figure 3.11, that the impact on catchment yield and hydrology would be more marked. Modelled monthly flow (DPIW 2008a, DPIW 2008b) for these sub-catchments is provided in Figure 3.14. Table 3.7 shows the breakdown of dam capacity against average annual and 'winter flows' (May to November inclusive) for these catchments.

Some insight to the impact to the natural flow regime for sub-catchments with high dam density is revealed by studies into the Hurst Creek sub-catchment within the Great Forester – Brid catchment. Of the 67 dams within the Hurst Creek catchment, 50 to 80% are not designed to allow the passage flows when they are below maximum flood level (DPIW 2007c). As a result upstream storages need to fill before flow can cascade downstream. This generally occurs between March and July. During summer most of the surface water is captured and the cascading effect delays the provision of winter flows to downstream reaches which contain significant natural values. An investigation into the effect of increasing the capacity of a farm dam from 22 ML to 37 ML estimated that overflow would be delayed on average by 2.2 days (Sinclair Knight Merz 2004). Given the number of dams that withhold flow, the cumulative effect on the delivery of flow downstream is likely to be significant.

Table 3.7 Modeled flow and dam capacity, cluster sub-catchments.

Cluster	Average Annual Flow (ML)	Total Licensed Dam Capacity	Capacity as% of Annual Flow	Capacity as% of May to November Flow	Capacity as% of May/June Flow
1	32091	1360	3.8	5	24
2	65450	4047	1.9	8.4	26
3	13433	544	4	5.2	23

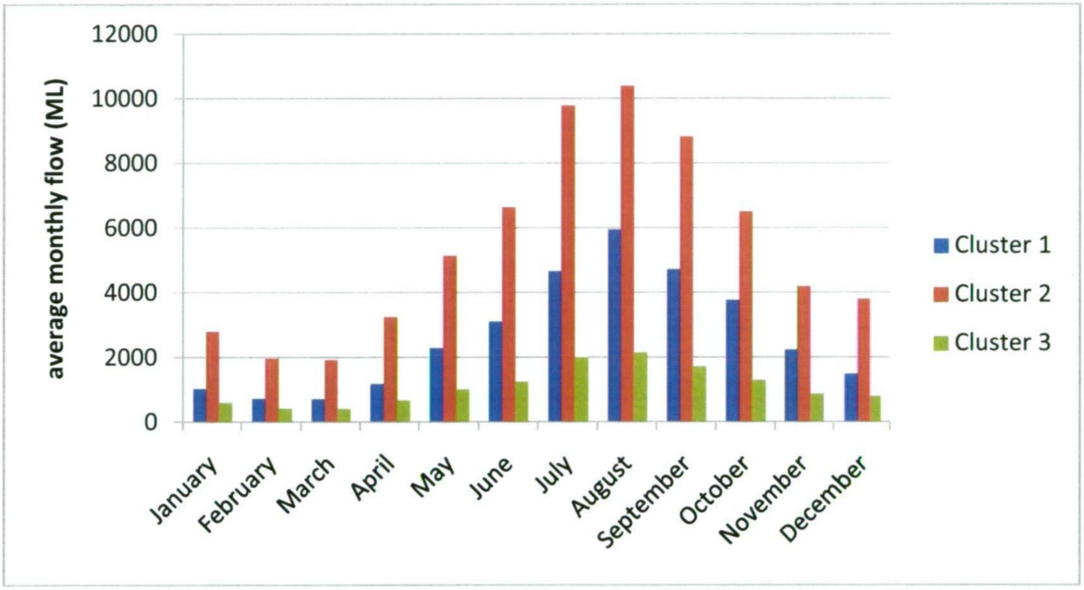


Figure 3.14 Combined average monthly flows for cluster sub-catchments.

3.2.2.3 Conservation of Freshwater Ecosystem Values

The Conservation of Freshwater Ecosystem Values (CFEV) project is an analysis of freshwater ecological values across Tasmania based on the comprehensive, adequate and representative (CAR) approach applied to the terrestrial reserve system (DPIW 2008c). CFEV is essentially an inventory of freshwater values within Tasmania based on existing environmental and ecological data. The CFEV database provides an assessment tool for management of freshwater systems.

A number of conservation management priority rankings have been attributed in CFEV to freshwater features such as river sections and wetlands. For the purposes of this analysis the conservation management ranking attribute used from the CFEV database is the 'CMPP2', as it is specifically recommended for the assessment of dams (DPIW 2008c, iii) and forms a component of the dam assessment process. A High or Very High CMPP2 rating 'highlights those freshwater-dependent ecosystems which have a high priority for active conservation management in the situation where future development and/or changes to land, water or vegetation management are proposed within the catchment, which may contribute to a change in aquatic ecological condition or status' (DPIW 2008c iii). The process for deriving this ranking is shown in Figure 3.13. Detailed descriptions of this attribute and others used in this analysis are provided in DPIW (2008c) and DPIW (2008d). The distribution of CMPP2 ratings for the Great Forester – Brid catchment is shown in Figure 3.16.

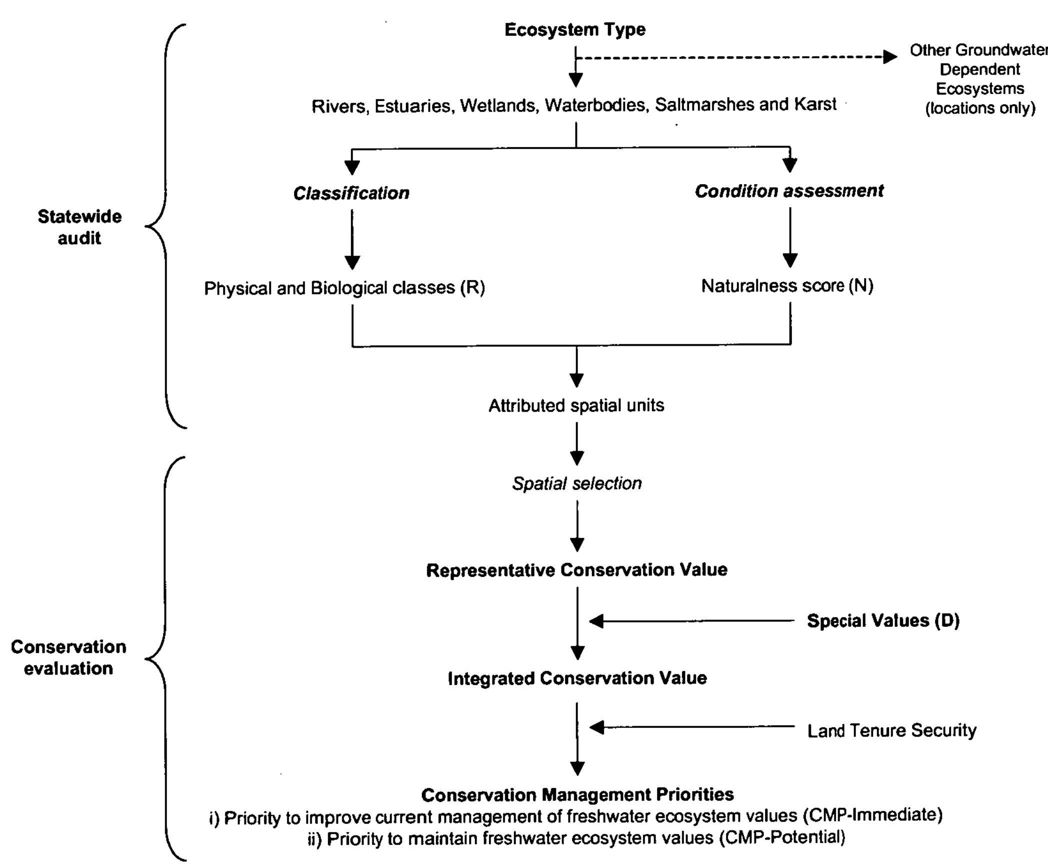


Figure 3.15 The CFEV assessment framework. Source: DPIW (2008c).

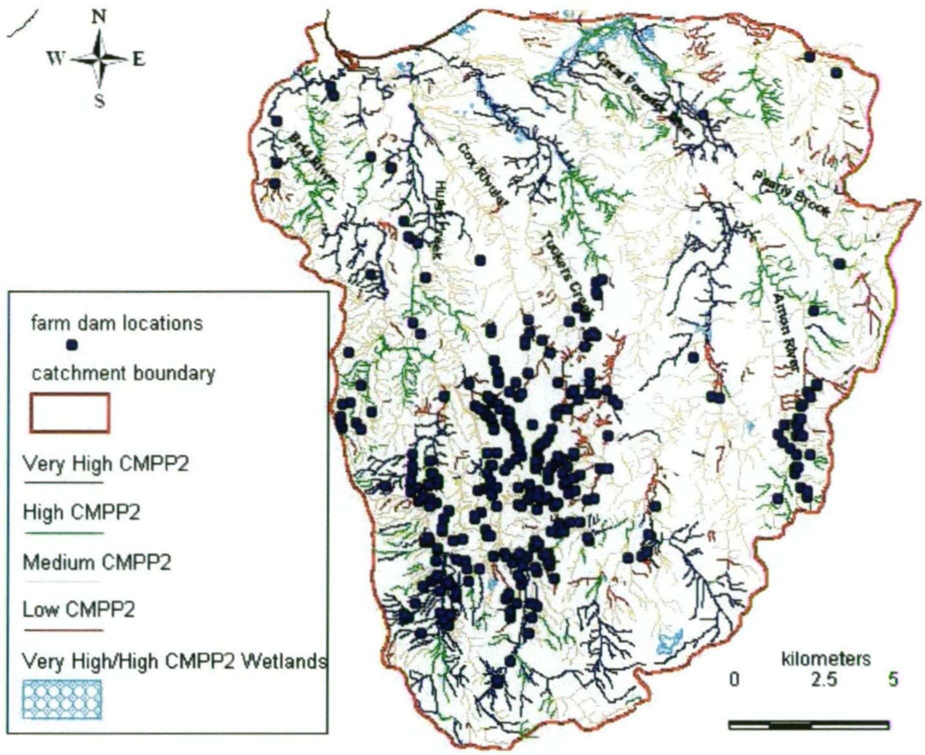


Figure 3.16 Great Forester – Brid catchment CFEV CMPP2 ratings and farm dam locations.

Across Tasmania, 20.4% and 26.2% of CFEV river sections are assessed as Very High and High CMPP2 respectively (DPIW 2008c). In the Great Forester – Brid catchment, this is 19% and 12.9%. Of the Very High CMPP2 river sections in the catchment, 14% contain a farm dam.

To some extent the determination of CMPP2 values for river sections that contain farm dams may be confounded by the use of a naturalness index. This index may reduce the CMPP2 value despite the potential presence of valued ecosystem components. For farm dams the surrounding land use is likely to be a factor in reducing the assessment in CFEV of naturalness. The Integrated Conservation Value (ICV) is independent of naturalness. It combines an assessment of representativeness with information on special values such as threatened species. The High and Very High categories can be used to flag locations that contain rare biological or physical classes, special values or both (DPIW 2008c). Across Tasmania, 1.6% of river sections are Very High ICV and 19.5% High ICV and these percentages are also apparent in the Great Forester – Brid catchment. Of the Very High and High ICV river sections in the catchment, 5% of both categories contain farm dams. The distribution of ICV categories across the catchment is shown in Figure 3.17.

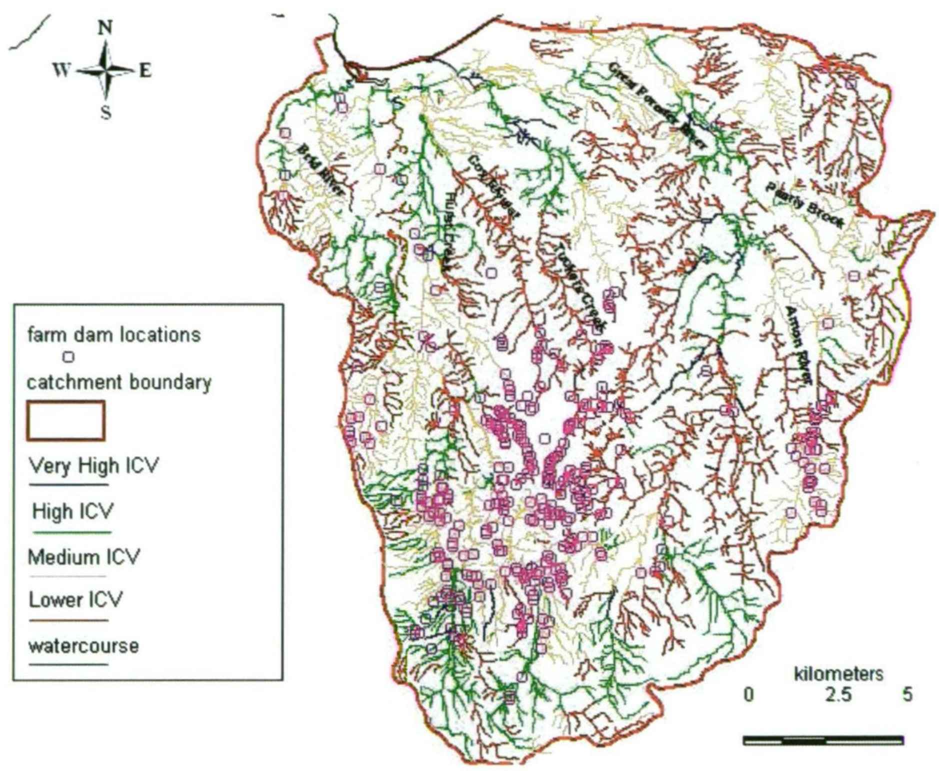


Figure 3.17 CFEV ICV categories Great Forester – Brid catchment.

3.2.2.4 River Fragmentation

Based on the CFEV river section data layer, 697 km of river section in the Great Forester – Brid catchment occurs upstream of one or more dams. This represents 34% of the total river length in the catchment. For river sections with a Very High or High CMPP2, 19.5% occur upstream of a farm dam while for Very High and High ICV river sections it is 16.9%. The proportion of first order streams (54.6%) and second order streams (25.4%) from river sections that are upstream of one or more farm dams is similar to that of the catchment as a whole. The locations of fragmented river sections are shown in Figure 3.18. The total stream length lost to farm dams is not readily known as dam polygons for GIS analysis are unavailable. It should be noted that this analysis does not include the stream gauge weirs on the Brid and Great Forester Rivers. The former is a significant barrier, while the latter is less restrictive, particularly for some species and in particular flows. (Plates 3.5 and 3.6).

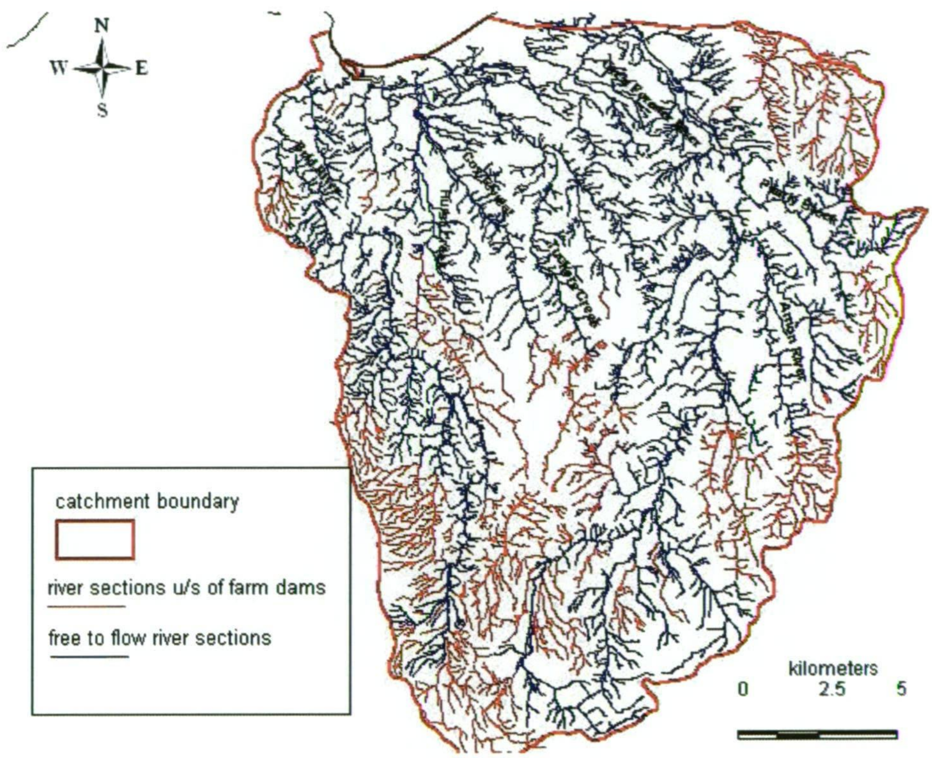


Figure 3.18 Location of river sections upstream of farm dam locations.

Plate 3.5 Brid River stream gauge weir.

Plate 3.6 Great Forester (2km u/s Forester Road) stream gauge weir.

3.2.2.5 Loss of Special Values

The Great Forester – Brid catchment contains a number of species and ecosystems of particular value. Two endemic species of burrowing crayfish, the Mt. Arthur burrowing crayfish (*Engaeus orramakunna*) and Scottsdale burrowing crayfish (*Engaeus spinicaudatus*) occur within the catchment. The Mt. Arthur burrowing crayfish is listed as vulnerable under both the Tasmanian *Threatened Species Protection Act* 1995 and the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*. The Scottsdale burrowing crayfish is listed as endangered under both Acts. The Scottsdale burrowing crayfish occurs only within the Great Forester – Brid catchment. The Mt. Arthur burrowing crayfish occurs across the Little Forester, North Esk and Pipers River catchment. Recent work by Wapstra et al. (2006) has expanded the previously described habitat of the Scottsdale burrowing crayfish suggesting that previous impacts may not have been identified. Both species occur in wet muddy areas and seepages, typically in gullies and other drainage lines that may be subject to dam development (Doran 2000). Doran (2000) has argued that dam construction is a threatening process. The timing and extent of seasonal high flows, native riparian vegetation, soil moisture and groundwater levels have been identified as particularly important for both the species (DPIW 2007d).

The giant freshwater lobster (*Astacopsis gouldi*) is endemic to Tasmania and occurs across the catchments of the north west and north east between the Arthur and Ringarooma Rivers (Threatened Species Section 2006). They are found throughout the Great Forester – Brid catchment. It is listed as vulnerable under both the Tasmanian *Threatened Species Protection Act* 1995 and the Commonwealth *Environment Protection and Biodiversity Conservation Act 1999*. Giant freshwater lobster are found in flowing and still water and occur in all stream sizes. They may be found in impoundments, however adults will not tolerate temperatures above 18° C and ideally require shade, an intact riparian zone, instream woody debris and a heterogeneous instream habitat. Juveniles are known to occur in headwater streams and Davies and Cook (2004) have suggested that this habitat is of particular importance to the species. Recolonisation of impacted habitat is slow, indicating a naturally slow dispersal but also possibly a lack of connectivity and suppressed populations (Threatened Species Section 2006). Instream farm dams have been identified as a barrier to lobster movement as well as changing flow, thermal regime and fluvio-geomorphology. Populations have declined in both the Brid and Great Forester catchments. Sedimentation in the middle and lower reaches of the Great Forester River has significantly reduced the available habitat (DPIW 2006).

These species could be used to assess cumulative impacts within the catchment and also across catchments. The known observations of these species and the known and potential range of the Scottsdale burrowing crayfish in relation to farm dam locations are shown in Figure 3.19. Across the catchment, 61 and 132 dams occur within the potential range of the Mt Arthur and Scottsdale burrowing crayfish respectively. The average dam density for the catchments that constitute the natural range of the giant freshwater lobster is 43 dams/100km^2. Dam density is particularly high across the catchments of northwest Tasmania which are particularly important for the species.

The potential for these dams to cumulatively impact on the burrowing crayfish species can begin to be determined by examining the most recent dam approvals within their known range. Eleven of the most recent dam approvals within the range of either species were examined. Five of the proposed dams were considered not to impact upon habitat likely to support the species. This conclusion, determined as part of the dam assessment process, is based on site descriptions and photographs, known locations and habitat preferences. For six of the proposals a recommendation was made to the Assessment Committee for Dam Construction (ACDC), the regulatory body that assesses farm dam proposals, that a survey be undertaken. Three of these were for both species, with two surveys for the Mt Arthur burrowing crayfish and one for the Scottsdale burrowing crayfish. In two instances the recommendations were rejected. In one case it was considered that even if the species was present that there were larger areas of habitat in the immediate area and therefore the impact would not be significant (ACDC 2009). In the second instance it was considered that given the distance from known observations a survey was not required and that if present the dam would not have a significant impact on the species (ACDC 2009).

In one instance Mt Arthur burrowing crayfish were identified as occurring both within the proposed area of inundation and in the surrounds. The dam was approved with the remaining habitat to be fenced as an offset. The lost population was estimated at 20 individuals although estimates were difficult due to felled vegetation. A second site was surveyed for both species; in this instance ***Engaeus mairener*** populations were identified. This burrowing crayfish species is endemic to north eastern Tasmania but not considered threatened and the proposal was approved without change. A single survey for both species was conducted for the remaining two proposals which were in close proximity on the same stream. An estimated 500 borrows were identified within the inundation areas. The species were not identified. As no threatened species were identified both dams were approved (ACDC 2008). The survey also noted burrows further upstream in an area likely to be the subject of a Forest Practices Plan (FPP).

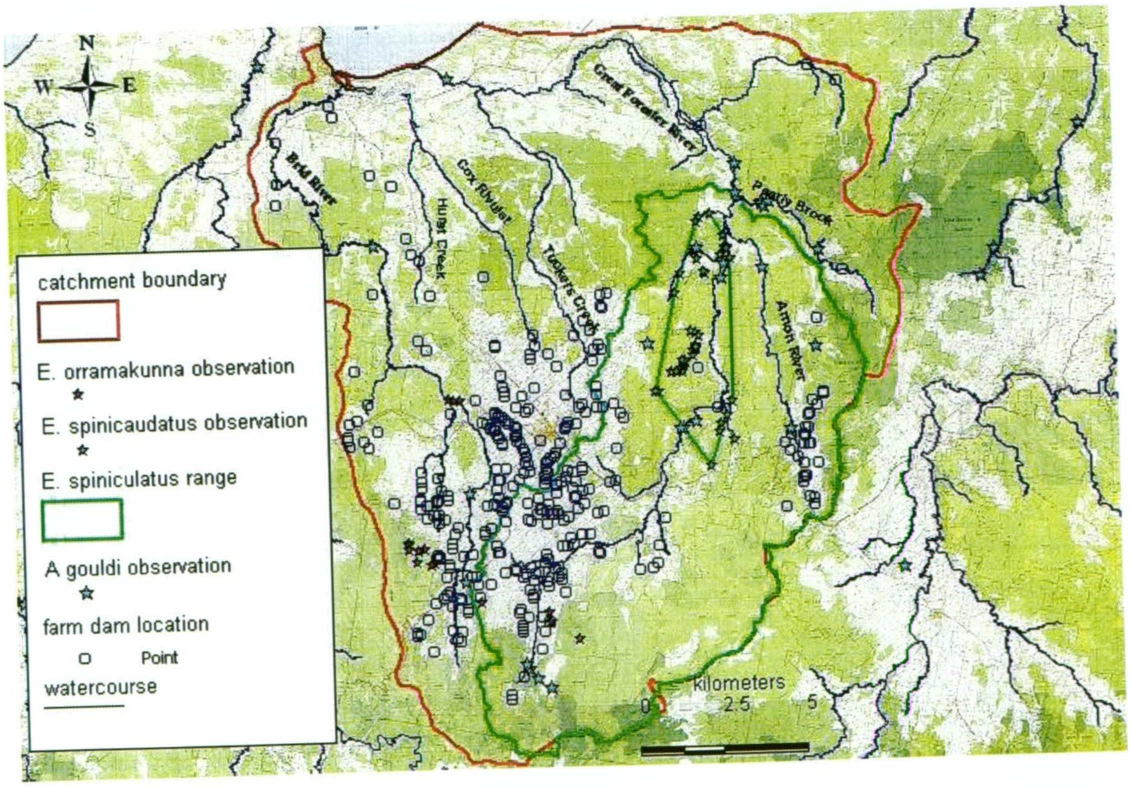


Figure 3.19 Mt. Arthur and Scottsdale burrowing crayfish and giant freshwater lobster observations: Great Forester – Brid catchment. Source DPIPWE (2010b).

3.2.3 Forestry

Between 2000 and August 2010, 206 FPP's were certified within the Great Forester – Brid catchment for harvesting operations (Forest Practices Authority 2010). Approximately half of the FPP are for establishment of hardwood and softwood plantations. Approximately half were established following the clearfelling of existing plantation and the remainder from native vegetation. Thirty one FPP were for the conversion of either forest or plantations to cleared land, while 19 were for the establishment of plantations on previously cleared land. A broad breakdown of areas for clearfell operations is provided in Table 3.8. Total plantation area (Figure 3.21) based on TASVEG data within the catchment is 98 km^2 (DPIW 2009a), with the Tasmanian Planning Commission (2009) estimating the total percentage of plantation cover in the catchment at 14%. The majority of forestry operations are in the headwaters streams of the catchment. The points shown in Figure 3.20 are the centroids of the FPP areas. Where this point is within 100 m of a watercourse, 68% of those watercourses are first order headwater streams.

Table 3.8 Vegetation types for clearfell and partial harvest operations in the Great Forester – Brid catchment 2000-August 2010. Source: Forest Practices Authority (2010).

Vegetation Type	Converted from (ha)	Converted to (ha)
Native vegetation	3339	1530
Hardwood plantation	673	2614
Softwood plantation	1621	2026
Non forest	1174	618

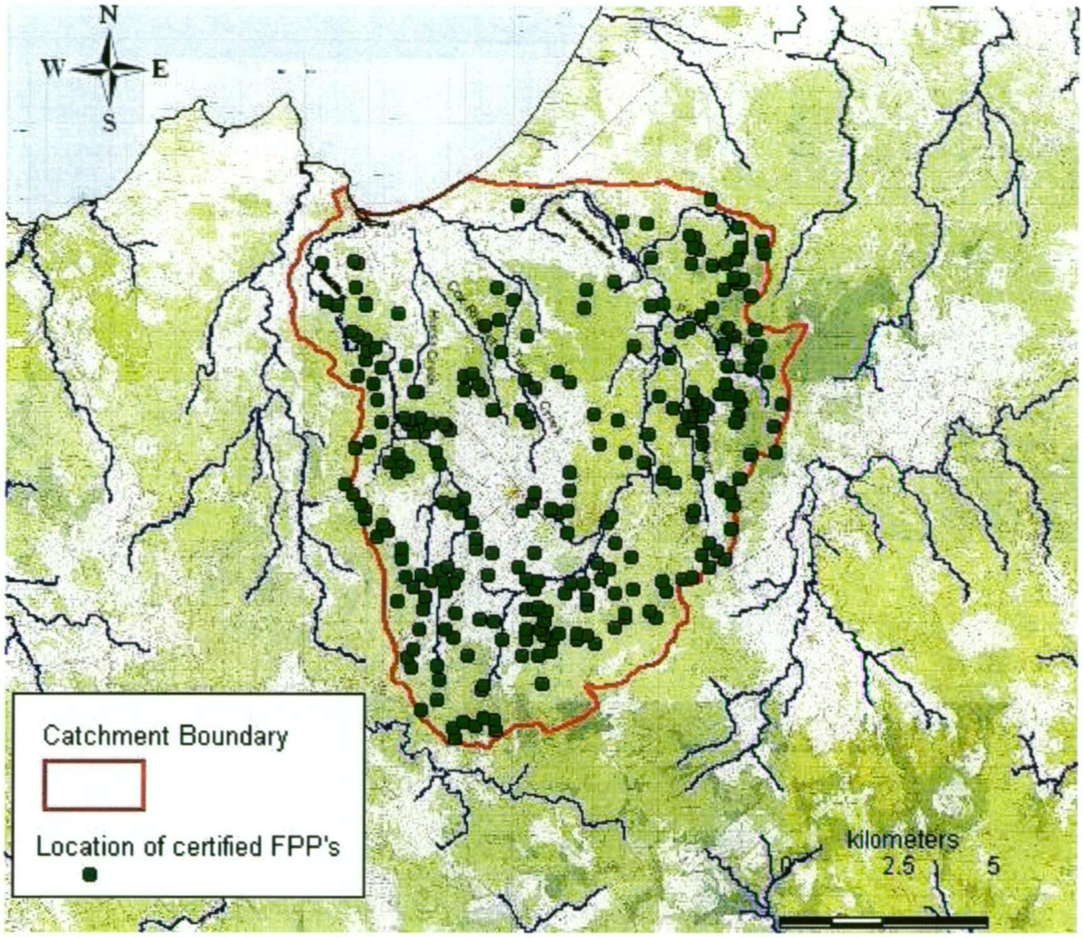


Figure 3.20 Location of certified FPP Great Forester – Brid catchment. Source: Forest Practices Authority (2010).

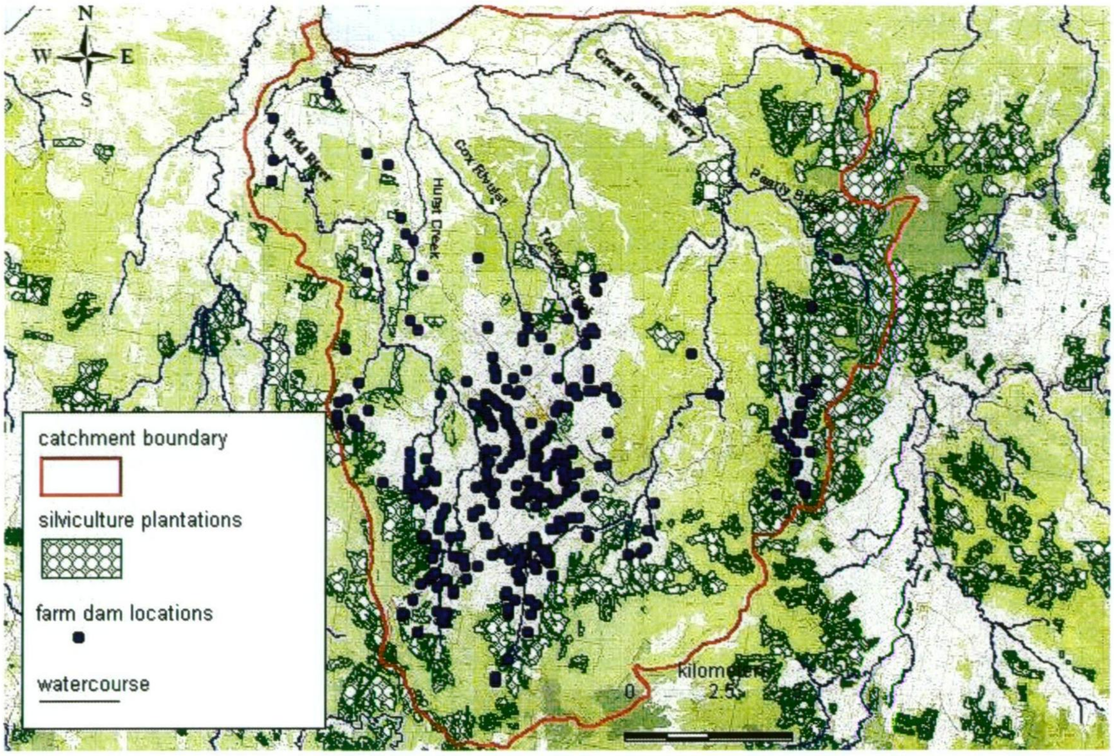


Figure 3.21 Plantations Great Forester – Brid catchment. Source: DPIW (2009a).

3.2.4 Water Licenses

The location of off-takes within the catchment tends to mirrors that of dam locations (Figure 3.22) and are associated with agricultural land uses. Total allocations are 24404 ML. The majority of allocations (22416 ML) can be extracted between November and April inclusive. There is currently a moratorium on takes during this period, all additional allocations are to be allocated outside this period.

Currently extraction points are not metered and therefore actual usage is not known. It is also assumed that there is some illegal extraction of water from the system (Ling et al. 2009). Exceedance rates are estimated at between 2 and 6 times the allocated amount, with three times the allocated amount considered to the average rate across all catchments (DPIW 2008b). Allocations are provided at varying sureties or reliabilities (Table 3.9). In the Great Forester – Brid catchment under historical conditions, sureties of 4 or less are fully met while those of 5 and above are met for

89% of the allocation (Ling et al. 2009). The majority of the allocations in the Great Forester – Brid catchment are surety 5 or 6.

Table 3.9 Department of Primary Industries, Parks, Water and Environment surety descriptions. Source DPIPWE (2010c).

Surety Description
High priority
1 Rights for the taking of water for domestic purposes, consumption by livestock or firefighting under Part 5 of the *Water Management Act 1999* and rights of councils to take water under Part 6 of the Act. Surety 1 water is expected to be available at about 95 percent reliability.
2 The water provision allocated to supply the needs of ecosystems dependent on the water resource.
3 Rights of licensees granted a water license as a replacement of the 'prescriptive rights' ('pre-Hydro Tasmania rights') granted under the previous Water Act 1957.
4 Rights of special licensees such as Hydro Tasmania.
Low priority
5 Rights issued for the taking of water otherwise than for the purposes described above under surety levels 1 to 4. This includes rights issued for the taking of water under Part 6 of the Act for direct extraction, and for winter storage in dams, for use for irrigation or other commercial purposes. Surety 5 water is expected to be available at about 80 percent reliability.
6 Rights at this surety level issued for the taking of water under Part 6 of the Act for direct extraction for use for irrigation and other commercial purposes and for winter storage in dams. Surety 6 water is expected to be available at less than 80 percent reliability.
7, 8 Water allocations available with a lower level of reliability than a surety 6 allocation.

Extractions account for 19% of total catchment yield for October to March inclusive (Ling et al. 2009). The impact of water extraction is most evident during periods of low flow (Viney et al. 2009). Graham et al. (2009) identify a measurable decline in the hydrological health of the catchment, particularly under recent climate. This has been attributed to alterations to the natural flow regime resulting from extractions and storages. The cluster sub-catchments employed to examine the impact of farm dams at a smaller spatial scale can also be employed to examine allocations. Allocations within Cluster 1 constitute 44.8% of the flow of the sub-catchment for the months November to April inclusive. For Cluster 2 this figure is 46% and for Cluster 3 it is 49%.

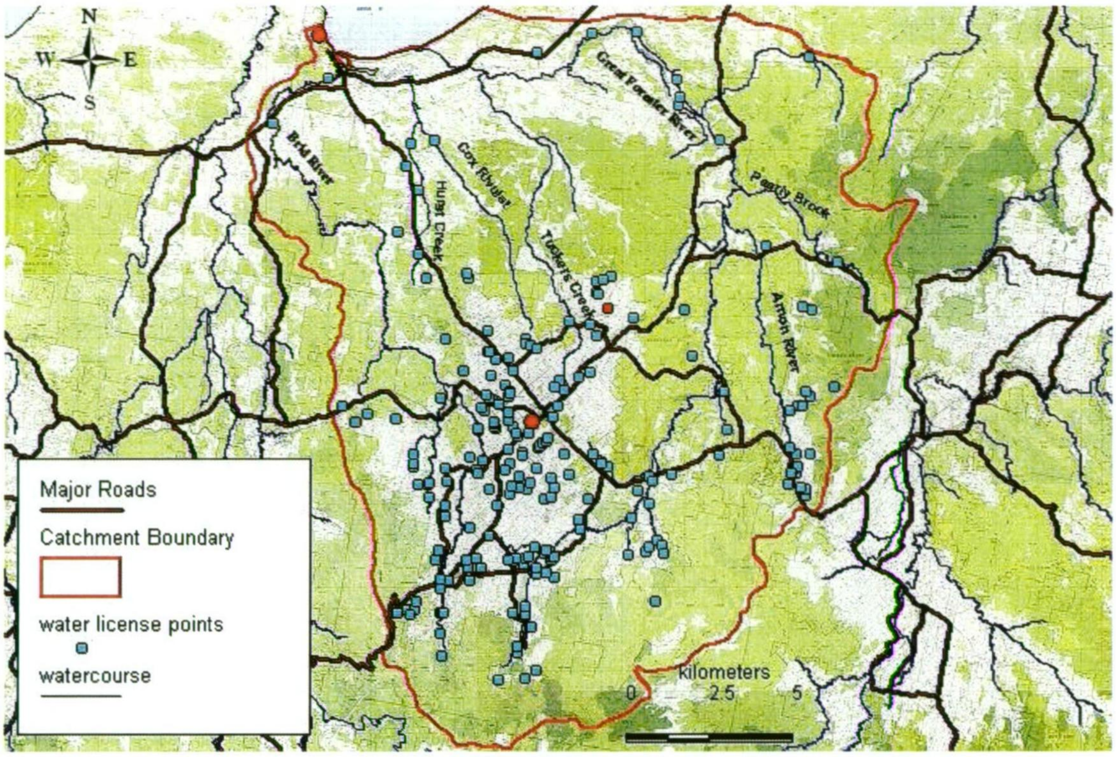


Figure 3.22 Locations of water license off-takes Great Forester – Brid catchment. Source: DPIPWE (2010a).

3.2.5 Future Development

While dam approval rates across Tasmania and within the Great Forester – Brid catchment fluctuate annually, the overall rate of approvals since the implementation of the *Water Management Act 1999* remains steady (Table 3.10). From 2003, numbers of smaller dams approved across Tasmania has dropped, while the number of large dam approvals (>1000 ML) has increased (Tasmanian Planning Commission 2009). This trend is continuing with an average dam capacity approved in 2008-09 of 229 ML compared to 165 ML in the previous year and 68 ML in 2000 (DPIW 2009b). Two dams for irrigation purposes contribute 40% of the total approved capacity of 5979 ML within the Great Forester – Brid catchment since 2000. Since 2001 there has been an increase in the percentage of instream dams approved in Tasmania in comparison with off stream. In 2005-06 83% of dam approvals were for instream dams. In the Great Forester – Brid catchment 47 of the 48 dams approved since 2000 are instream dams. It should be noted that these figures are derived from dam data from the Water Information System Tasmania for which an approval date is provided.

There is no indication that these trends are likely to change into the near future. The approval of dam works permits is an 'effectiveness indicator' for the Department of Primary Industries, Parks, Water and Environment, with a statewide target for 2009-10 of 100 new dam approvals (DPIW 2009c). For 2007-08 137 dam applications were received, with 110 received the following year. All these applications resulted in a dam approval being granted (DPIW 2008e). In 2008-09 25219 ML of capacity was approved, 24475 ML of this for irrigation purposes. Average annual approved capacity since 2000 is 20227.8 ML. If the implementation of the large irrigation dams proposed for the catchment by the Tasmanian Irrigation Development Board is successful, these proposals, which are discussed below, may alleviate the demand for further on farm instream irrigation storage. These dams, however, are all significant, large instream dams, some of which are in the lower middle catchment and on the main river.

Table 3.10 Dam approvals 2000-2009 Tasmania and Great Forester – Brid catchment.

Year	Statewide Dam Approvals	Capacity (ML)	Great Forester – Brid Dam Approvals	Capacity (ML)
2000	197	9610	4	260
2001	158	12922	5	408
2002	179	21556	2	110
203	175	21226	6	183
2004	198	24810	6	145
2005	158	18274	3	43
2006	91	20223	7	1558
2007	131	25052	8	491
2008	141	28115	5	2692
2009	86	20995	2	90
Total	1514	202783	48	5979

Currently there are four major instream irrigation dams within the Great Forester – Brid catchment proposed by the Tasmanian Irrigation Development Board (Figure 3.23). They are proposed for the Brid River (3850 ML), the middle reaches of the

Great Forester River downstream of the Cluster 2 sub-catchment (8500 ML) ,Oxberry Creek in the lower Great Forester catchment (8947 ML) and Parrs Rivulet, a tributary of the middle reaches of the Great Forester River (6500 ML) (Ling et al. 2009). The smaller Headquarters Road dam (1980 ML) in the upper Great Forester River was approved in 2008.

Predicting the future trends for forestry within the catchment is problematic due to the volatility of the industry (Viney et al. 2009). Ling et al. (2009) suggest an increase in plantations across north eastern Tasmania of 147 km^2 and an increase in total forest cover from 25% to 27% across the region (Figure 3.23). This trend has also been identified from the long term increases in plantations, particularly hardwood (Tasmanian Planning Commission 2009). There is also a trend towards a younger overall plantation age across the state as shown in Figure 3.24 (Tasmanian Planning Commission 2009). The greatest increases within the Great Forester – Brid catchment are predicted for the middle and lower reaches of the Brid River and the lower reaches of the Great Forester River reflecting establishment of plantations on private land. Current industry trends suggest a move away from native forest logging and it is possible that there may be a reduction of forest activity in the upper reaches of the catchment.

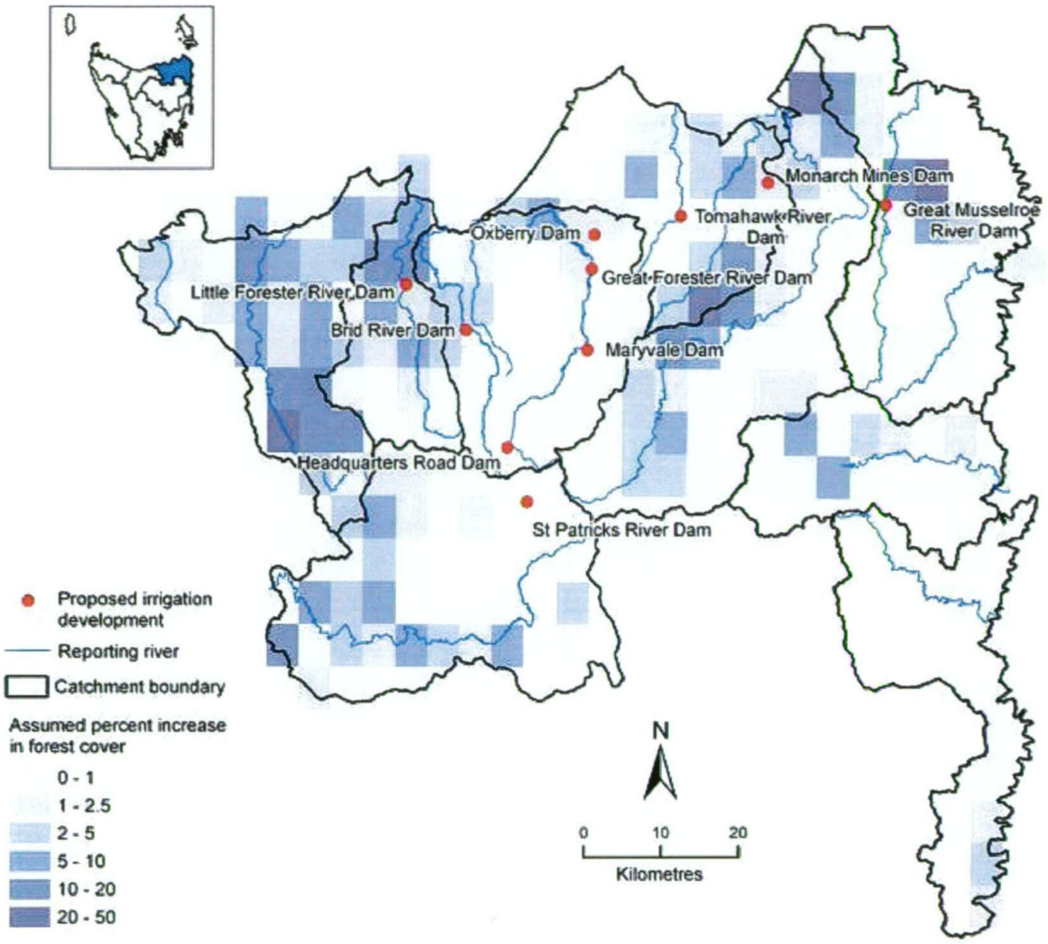


Figure 3.23 Predicted future plantation increase and locations of proposed irrigation dams. Source: Ling et al. (2007).

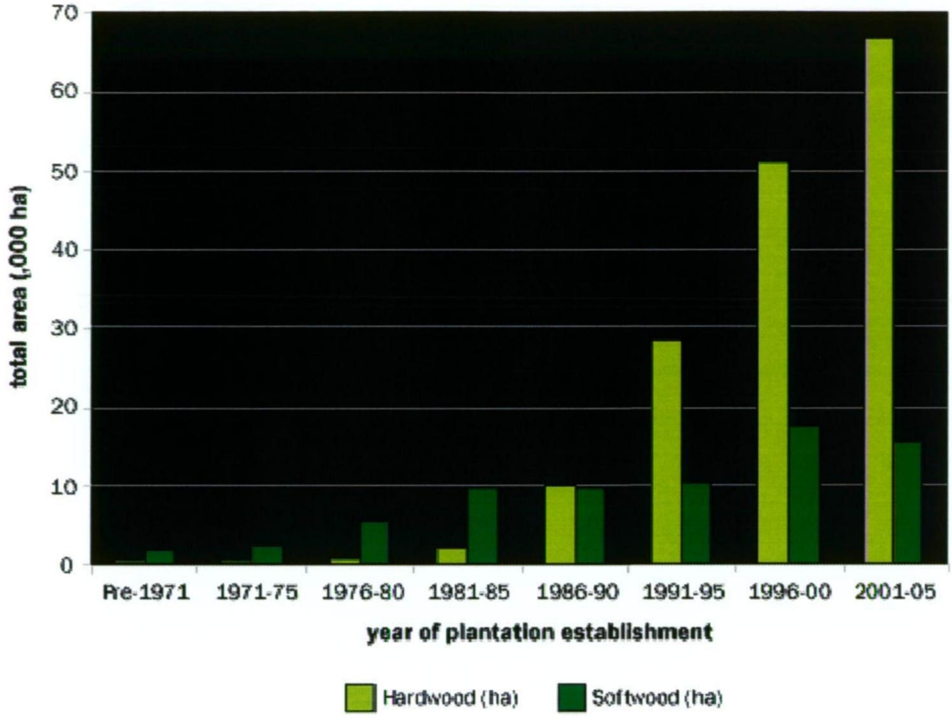


Figure 3.24 Age of plantations across Tasmania as at 31 December 2005. Source: Tasmanian Planning Commission (2009).

Consideration of current and future cumulative impacts from a catchment perspective examined thus far have utilised data that reflects historical conditions. Under either projected climate change and/or natural climate variability, these impacts may differ in effect and magnitude. Data for recent climate in Tasmania (1997-2007) shows that this period has been characterised by lower than average rainfall with a 6% reduction across the state in comparison to the historical climate (1924-2007) (CSIRO 2009b). The greatest reduction is in the north east of the state with a reduction of 12%. These reductions are not distributed evenly across seasons. The largest reductions are in autumn (maximum reduction of 21% for the north east) and summer (12% reduction for north east). Winter rainfall has also been considerably less (14% reduction) for north east. These trends are significant for assessing the impact of dams, particularly the reduction in autumn and winter flows. For the Great Forester – Brid catchment total average catchment yield for the recent climate is 189.6 GL, a reduction of 26% from the historical average (CSIRO 2009b).

Recent analysis of potential future climate in Tasmania (CSIRO 2009b) considered three projections: wet extreme, median and dry extreme. The implications for runoff for the modelled climate projections are shown in Table 3.11. Runoff is expected to decrease in north eastern Tasmania with the greatest reductions across the State expected to occur in summer, autumn and spring.

Table 3.11 Change in annual and seasonal runoff under the future climate relative to historical climate. Source: CSIRO (2009b).

	Annual	Summer	Autumn	Winter	Spring
Wet extreme future climate					
Change in runoff mean	2%	7%	1%	2%	−1%
Percentage of Tasmania with decreasing runoff	28%	22%	28%	32%	49%
Median future climate					
Change in runoff mean	−3%	−6%	−4%	−1%	−6%
Percentage of Tasmania with decreasing runoff	82%	86%	80%	57%	83%
Dry extreme future climate					
Change in runoff mean	−8%	−18%	−9%	−1%	−14%
Percentage of Tasmania with decreasing runoff	97%	92%	91%	50%	94%

Within the Great Forester – Brid catchment average annual catchment yield is expected to decrease under all scenarios. Table 3.12 provides a summary of predicted changes to catchment runoff under future climate projections and future development in the Great Forester – Brid catchment. Future development is limited to projected increases in plantation forestry and large irrigation projects and it does not account for increases in extractions or storage in farm dams. Peak flows are also predicted to change under climate projections (Table 3.13) with further changes, including some potentially significant reductions in peak flow, expected depending on the final configuration of any approved large scale irrigation developments. The projected changes provide a broad indication of possible changes and it is likely that there will be some variability across seasons and years and spatially across the catchment depending on the distribution of catchment land use.

Table 3.12 Projected changes to catchment runoff in the Great Forester- Brid catchment under future climate change and future development. Source: CSIRO (2009b).

Historical	Wet Extreme	Median	Dry Extreme
total mean annual flow (GL)			
256	247.7	234.6	219.4
Total flow October -March			
83.1	81.5	73.3	65.7
Percentage of water extracted from October - March total flow			
19	19	20	21
Percentage change to catchment runoff under future development			
-	-1	-1	-1

Table 3.13 Changes to peak flows in the Great Forester – Brid catchment under projected climate change and future development.

Peak Flow	Wet extreme	Median	Dry extreme	% change relative to wet extreme	% change relative to median	% change relative to dry extreme
2 year (ML/day)	4355	4226	3943	-4.06	-10.70	-5.71
5 year (ML/day)	6242	6106	5764	-2.57	-3.28	-5.99
10 year (ML/day)	7835	7742	7357	-2.05	-2.15	-3.3

3.2.6 Summary

The most readily identifiable cumulative impact of farm dams in the Great Forester – Brid catchment is the reduction of natural flow of the total combined storage capacity. This impact is evident from a whole of catchment perspective as a reduction in catchment yield. The significance of that reduction on the catchment ecosystem as a whole or on particular elements within it is not fully understood. Furthermore, the change to the natural flow is temporally and spatially variable. An understanding of the cumulative impact of farm dams on ecosystem processes needs to be framed within a scope that accounts for that variability. It is apparent within the Great Forester – Brid catchment that the greatest impact to the natural flow regime is likely to occur following the first significant rainfalls in autumn and early winter. This phenomenon is likely to be evident across most catchments in Tasmania with significant dam development. The concentration of farm dams in particular areas is also an important consideration in the determination of cumulative impacts, particularly as these areas are subject to impacts from intensive agriculture. Critically, farm dam usage is not known and the actual volume of water lost from the catchment cannot be quantified.

Farm dams are known to have a range of other cumulative impacts that are more complex in nature. This includes alteration of sediment regimes and instream geomorphological processes, impacts to water quality and fragmentation of river systems. The total numbers of dams and the proportion of the catchments river sections that are upstream of at least one farm dam suggest that these impacts are likely to be evident and may be significant at particular spatial and temporal scales. It is also apparent from the examination of the impact of recent dam approvals on the threatened burrowing crayfish species that dam construction is one of a number of possible cumulative impacts on these species. There are other natural values that may also be of interest in this regard, for example lowland wetlands, threatened riparian flora and threatened native vegetation communities.

Water licenses can be assessed in much the same way farm dams with the impact occurring predominantly in summer and early autumn. As with dams, extractions are currently largely unmetered, and therefore usage both in terms of total extraction and the timing of extractions is unknown. The impact on catchment flow during the period of allowable extraction is certainly significant but there is considerable spatial variability with the greatest impact on the natural flow regime occurring during low flows in areas already subject to a number of other stressors.

The impact of forestry, particularly plantation establishment, on catchment yield is more difficult to assess. However the literature provides a strong case for the inclusion of the impact of forestry on yield in any determination of the sustainable use of natural resources within a catchment. There is at least a sufficient understanding to quantify the impact of forestry operations on catchment yield in a meaningful way. There is also strong evidence that forestry operations, despite management protocols, impact directly on aquatic ecological processes. This is particularly evident for headwater streams. Within the Great Forester –Brid catchment there is significant forestry activity and plantation development and it is expected that this will be ongoing. The majority of this activity is located within the headwaters of the catchment.

While it is clear that each of these regulatory processes results in cumulative impacts on the catchment, it can be expected those impacts will interact in a number of different ways resulting in an overall cumulative impact at the catchment and sub-catchment scale. The natural flow regime is the key variable impacted by the combined cumulative effect of these multiple regulatory decisions. It drives aquatic ecosystem processes and there are identifiable direct effects on aquatic biota of all these regulatory processes, including on the same ecosystem elements. Furthermore these same processes are played out in a similar manner across many other catchments in Tasmania as the levels of development within the Great Forester – Brid catchment are indicative of many Tasmanian catchments outside of the western portion of the state. Consequently many natural values of aquatic ecosystems are subject to the same cumulative pressures across their natural range. The importance of addressing the cumulative impact of these decisions making processes is highlighted by the evidence that future development is likely to continue at the current rate at least. The addition of large scale irrigation proposals represents a significant element in the possible trajectory of future development. The extra dimension of climate change acting on the cumulative effects of these regulatory decisions adds further urgency to the development of appropriate mechanisms to ensure that there cumulative impacts are not resulting unsustainable development of natural resources.

The following chapter examines the regulatory and policy environment that controls these activities. The extent to which cumulative impacts have been addressed within that environment is discussed. Consideration is given to the extent to which the current legislative and policy framework can accommodate an appropriate level of CEA and alternative approaches are suggested.

Chapter 4 Legislation, Policy and Resources

4.1 Introduction

This chapter examines the principal legislation governing farm dam construction, forest practices and water allocation in Tasmania. The application of legislation is considered in terms of the extent to which the cumulative impacts of those activities are addressed. The potential of current legislation to regulate cumulative impacts is also discussed. Similarly, associated key policies, plans and strategies are also reviewed.

Current resources that may have application in cumulative effects assessment of the impact at the catchment spatial scale of farm dams, water allocation and forest practices are considered. It is not intended to be an exhaustive review, rather key datasets from current monitoring activities and recent development of catchment models are the focus. Critical gaps in the resources required for the identification and assessment of cumulative impacts are discussed.

4.2 Legislation

4.2.1 Resource Management and Planning System

The Resource Management and Planning System (RMPS) was established in 1994. Its principle aim is to ensure sustainable use of Tasmania's natural resources through the realisation of the following objectives:

- promote the sustainable development of natural and physical resources and the maintenance of ecological processes and genetic diversity;
- provide for the fair, orderly and sustainable use and development of air, land and water;
- encourage public involvement in resource management and planning;

- facilitate economic development in accordance with the objectives set out above; and
- promote the sharing of responsibility for resource management and planning between the different spheres of Government, the community and industry in the State (Resource Planning and Development Commission 2003, 6).

Sustainable development as referred in these objectives is defined as 'managing the use, development and protection of natural and physical resources in a way, or at a rate, which enables people and communities to provide for their social, economic and cultural well-being and for their health and safety while:

- sustaining the potential of natural and physical resources to meet the reasonably foreseeable needs of future generations;
- safeguarding the life-supporting capacity of air, water, soil and ecosystems; and
- avoiding, remedying or mitigating any adverse effects of activities on the environment.' (Resource Planning and Development Commission 2003, 6).

These overarching objectives are enshrined in a number of pieces of legislation, including the *Water Management Act 1999*, which is the principle regulation for dam construction and water licensing. There are a number of other natural resource activities that are largely outside of the umbrella of the RMPS, including forest practices and mineral exploration. While the objectives are underpinned by concepts such as inter-generational equity, conservation of biodiversity, efficiency, the precautionary approach and strategic planning, there is no direct reference to cumulative effects within the RMPS.

4.2.2 State Policy and Projects Act 1993

The *State Policy and Projects Act 1993* allows for the establishment of State Policies within the RMPS. The intent of State Policies is to address long term resource development issues through the establishment of a consistent State wide approach consistent with the objectives of the RMPS. A State Policy is binding on any person, State Government agencies, public authorities and planning authorities and local government planning schemes are required to be consistent with State Policies (Resource Planning and Development Commission 2003). The State Policy currently relevant to farm dam, water license approvals or the cumulative impacts of forestry on catchment values is the State Policy on Water Quality Management 1997.

4.2.3 Water Management Act 1999

The *Water Management Act 1999* is compliant with the objectives of the RMPS and establishes a consistent system for the allocation of water licenses, establishes a dam approval process, provides for the development of Water Management Plans and establishes mechanisms for the provision of environmental flows. The approval process for farm dams and water licenses is considered further in Section 4.4.1.

The Act has the following additional objectives:

- promote sustainable use and facilitate economic development of water resources; and
- recognise and foster the significant social and economic benefits resulting from the sustainable use and development of water resources for the generation of hydro-electricity and for the supply of water for human consumption and commercial activities dependent on water; and
- maintain ecological processes and genetic diversity for aquatic and riparian ecosystems; and
- provide for the fair, orderly and efficient allocation of water resources to meet the community's needs; and
- increase the community's understanding of aquatic ecosystems and the need to use and manage water in a sustainable and cost-efficient manner; and

- encourage community involvement in water resource management (*Water Management Act 1999* Part 2 (6)).

Water Management Plans are established with the aim of providing a clear statement of objectives for the management of a particular water resource, usually a catchment. Plans provide for limits to allocations, environmental flows and monitoring and review and must include a statement of environmental objectives, a water regime to achieve those objectives and assessment of the ability of the Plan to meet those objectives (Water Management Act 1999, Part 4 Section 14). Currently in Tasmania there are six Water Management Plans in effect (DPIPWE 2010d), with the Great Forester Water Management Plan the first to be implemented in 2003.

The key elements of the Water Management Plan for the Great Forester River (DPIWE 2003) catchment is a restriction management protocol that prevents Surety 5 and Surety 6 direct takes during low flows, the provision of a minimum environmental flow and moratorium on Surety 5 summer takes. Under the Plan future expansion of irrigation is to be met by the additional storage of winter flows and extraction of summer high flows. Existing historical use in excess of licensed allocations is recognised as Surety 6 takes under the Plan. The Plan has a strong monitoring component but its overarching emphasis is on water allocation, including the minimum allocation to the environment.

The Act requires a license to take water from a watercourse, water body or dispersed surface water. Water licenses are granted by the Minister responsible for the Act, however DPIPWE has been delegated the authority to grant or refuse licenses in accordance with the Act. Licenses are granted to the applicant and are considered personal property, are not associated with land title, are subject to conditions that may be unique to that license and are required to conform to any relevant Water Management Plan. Water licenses are generally granted for forty years and can be sold or leased with approval. For properties that have a river frontage the Act allows the taking of water for stock or domestic use without a license. Regulations impose limits, for example 90 L a day per head of cattle (Water Management Regulation 2009, section 4).

The Act provides for the constitution of an Assessment Committee for Dam Construction (ACDC) responsible for the approval or refusal of individual dam permit applications (*Water Management Act 1999,* Section 138). The ACDC is required have regard to the objectives of the Act, however there is no specific requirement under the Act for it to consider any matters outside of individual dam applications. While there is provision for dams to be assessed under the *Environmental Management and Pollution Control Act 1994*, the ACDC approval powers extends to all dams. However, dam storages less than 100 ML, where no additional information has been recommended and where no representations are made, may be approved by DPIPWE under delegation. The ACDC does not determine the granting of the water licenses required for dams. In 2007, the enactment of the *Dam Works Legislation (Miscellaneous Amendments) Act 2007* exempted holders of dam works permits from requiring a threatened species permit under the *Threatened Species Protection Act* and exempted the requirement for an FPP under the *Forest Practices Act 1985* where a dam works permit is held. These changes required the ACDC to be responsible for all environmental considerations relating to dam applications (DPIW 2008e)

While the objectives of the Act and those of the RMPS refer to 'sustainable development' and the need to 'maintain ecological processes', these objectives are broad in nature and are balanced by others seeking to facilitate economic development. While arguably the need to manage cumulative effects is a requirement for realising any objective regarding sustainability, it has been shown that cumulative effects assessment, not only need to be explicitly cited as a regulatory responsibility, it also requires definitions within regulation in order to ensure that facets such as reasonably foreseeable future actions are addressed. Nevill (2003) suggests that the difficulty and importance of managing cumulative effects in water and catchment management has been seriously underestimated, a view shared by Davies (2001) in the Tasmanian context. A review of water management legislation (Maher et al. 2001) determined that while the *Water Management Act 1999* did not address cumulative effects, the only other water management legislation in other state jurisdictions to do so was in New South Wales. In this state, the *Water Management Act 2000* refers to 'the cumulative impacts of water management licenses and approvals and other activities on water sources and their dependent ecosystems should be considered and minimised' (Maher et al. 2001, 96).

While Water Management Plans may set minimum flows and restrictive management of takes at low flows they do not address cumulative effects in any explicit sense. While each plan is intended to provide a framework to realise the objectives of the Act it cannot be said that Water Management Plans fully address all the issues necessary to ensure an objective such as 'maintain ecological processes and genetic diversity for aquatic and riparian ecosystems' (*Water Management Act 1999* Part 2 Section 6 (c)). The Great Forester Water Management Plan, for example, does not consider the full environmental impact of farm dams. The objectives of Water Management Plans do not include a need to address cumulative effects.

Maher et al. (2001) and Neville (2003) consider that the only way to manage cumulative effects within catchments is to establish caps on development well in advance of development meeting those limits. The suggested mechanism for achieving this is integrated catchment management. These authors identify that the *Water Management Act 1999* has the capacity to establish mechanisms to develop strategic catchment based caps on development but only for allocation of water. Other development such as drains, farm dams, vegetation clearance and the flow on effects of projects such as large irrigation dams (vegetation clearance, impacts of land use change) also require a strategic approach including development constraints within an integrated catchment management approach. Bellamy et al. (2002) recognise the difficulty in achieving integrated catchment management in Tasmania in an environment of competing sectors. Maher et al. (2001) argues that catchment management legislation should have primacy over other resource management legislation as a best practice approach to catchment management.

4.2.4 Forest Practices Act 1985

The *Forest Practices Act 1985* is not part of the suite of legislation that is required to conform to the objectives of the RMPS. The Act established the Forest Practices Authority (FPA). The FPA is governed by a board appointed by the relevant Minister and administers the forest practices system and also provides policy advice to the relevant Minister. The FPA is required to further the objective of the forest practices system (*Forest Practices Act 1985*, Part 1A, Section 4B). That objective is to 'achieve sustainable management of Crown and private forests with due care for the environment while delivering, in a way that is as far as possible self-funding –

- an emphasis on self-regulation; and
- planning before forest operations; and
- delegated and decentralized approvals for forest practices plans and other forest practices matters; and
- a forest practices code which provides practical standards for forest management, timber harvesting and other forest operations; and
- an emphasis on consultation and education; and
  - an emphasis on research, review and continuing improvement; and
  - the conservation of threatened native vegetation communities; and
- provision for the rehabilitation of land in cases where the forest practices code is contravened; and
- an independent appeal process; and
- through the declaration of private timber reserves – a means by which private land holders are able to ensure the security of their forest resources.' (*Forest Practices Act 1985* Schedule 7).

The Act is largely an instrument to establish governance, operational procedure and regulatory definitions that govern the forest practices system.

Forest operations require the certification by the FPA of an FPP. FPP are required to conform to the Forest Practices Code, a requirement of the Act (Forest Practices Act 1985, Part IV). The code (Forest Practices Board 2000) contains management prescriptions for individual forest harvesting operations and associated developments such as access roads. Protection of natural values, including streams and watercourses, is largely determined within the code at the coupe level. The general principles within the code regarding protection of biodiversity call for a strategic approach through a systematic reserve system, in addition to management prescriptions, in order for the forest practices system to contribute to the conservation of biodiversity at a State and regional level (Forest Practices Board 2000, 51). The code is almost entirely restricted to site specific operational management prescriptions.

4.3 Policy, Plans and Programs

4.3.1 State Policy on Water Quality Management

The objectives of this policy, enacted through the *State Policy and Projects Act 1993*, are to;

- focus water quality management on the achievement of water quality objectives which will maintain or enhance water quality and further the objectives of Tasmania's Resource Management and Planning System;
- ensure that diffuse source and point source pollution does not prejudice the achievement of water quality objectives and that pollutants discharged to waterways are reduced as far as is reasonable and practical by the use of best practice environmental management;
- ensure that efficient and effective water quality monitoring programs are carried out and that the responsibility for monitoring is shared by those who

use and benefit from the resource, including polluters, who should bear an appropriate share of the costs arising from their activities, water resource managers and the community;

- facilitate and promote integrated catchment management through the achievement of objectives (a) to (c) above; and
- apply the precautionary principle (State Policy on Water Quality Management 1997, Section 6).

The Policy establishes protected environmental values, for example recreational waters, which are to be protected by the establishment of appropriate water quality guidelines. The most stringent set of guidelines are to be adopted under the Policy as water quality objectives. These objectives are seen as a measure of the success of the management strategies for point and diffuse pollution sources detailed in the Policy.

Management strategies under the Policy are broad in scope and are in the form of over arching principles. This is particularly the case for diffuse sources of pollution where the Policy relies heavily on the use of the term 'environmental best practice' and for the development of appropriate guidelines for various activities. For forestry operations the Policy requires forestry activities under the *Forest Practices Act 1985* to be conducted in accordance with the Forest Practices Code 2000. For water allocation and dam approvals the Policy requires that 'when issuing or reviewing water rights and other licenses or permits which allow water abstraction, diversion or the construction of in-stream impoundments, water management authorities must take account of the likely effects of the proposed action on water quality, and whether it will prejudice the achievement of water quality objectives' (State Policy on Water Quality Management 1997, Section 14). The Policy does not specifically refer to the impact cumulative effects on water quality.

The State Government is now proposing to convert the policy into an Environmental Protection Policy under the *Environmental Management and Pollution Control Act 1994* following a review (DPIPWE 2010e).The review acknowledged that no water quality objectives or guidelines have been identified and submissions to the review identified this as a significant barrier to the development of integrated catchment management (DPIPWE 2010e, 15).

4.3.2 Policies Associated with the *Water Management Act 1999*

A number of policies have been established under the *Water Management Act 1999* to aid in the implementation of the objectives of the Act. The Water for Ecosystems Policy (Water Management Policy 2001/1) provides a framework for implementing environmental flows. The *Water Management Act 1999* requires the incorporation of environmental flows in the daily management of catchments, however implementation of Water Management Plans to facilitate this is not a specific requirement and the policy is a response to this. Under the policy the environmental flow is to be an average flow on at most a monthly time step. The policy distinguishes between 'unstressed' and 'stressed' aquatic ecosystems. The Environmental Water Requirement is defined under policy as the water regime required to sustain ecological values at low level of risk. The Water Provision for the Environment is defined as that component of the Environmental Water Requirement that can be met. Stressed aquatic ecosystems are defined as exhibiting significant scientific evidence of degradation related to allocations, consumptive use and storage or where the full Environmental Water Requirement is not provided.

For unstressed aquatic ecosystems the Water Provisions for the Environment are to be equal to the Environmental Water Requirement with triggers for more detailed investigation once 'water allocations reach a preset level above which only the Water Provision for the Environment would be left in the waterbody' (*Water Management Act 1999* Policy 2001/1, D (5)). For stressed aquatic ecosystems a low flow period Environmental Water Requirement is to be established, while outside of this period an holistic flow based on an assessment of spawning flows, flushing flows and channel maintenance is to be implemented. Currently, three catchments have holistic environmental flows (DPIPWE 2010f). The policy acknowledges that in catchments with moderate or high over allocation or where significant development may impinge upon the Environmental Water Provision, a moderate risk Environmental Water

Provision may be allocated. This is an acknowledgement of the difficulties encountered when development caps are not set early and management is attempted after catchments become stressed.

Water Resources Policy 2003/1 consists of guidelines for the assessment of allocations from watercourses during winter, including water allocations for dams. The policy acknowledges that ideally, no new water allocations should be made until catchment scale sustainable water allocation limits have been derived. This limit is the maximum volume that can be extracted after consideration of the water regime required for the environment, including high and low flows for in stream riparian, wetland and estuarine environmental processes (Water Resources Policy 2003/1, 4). Because of the economic impact of such an approach an interim measure is provided under the policy. This measure determines the water available for allocations by considering the volume of water available at 80 % reliability after the environmental water allocation and existing allocations are removed.

The policy acknowledges the lack of information on existing levels of abstraction and other water uses and the impact of water development on water dependent ecosystems such as instream habitat, riparian vegetation, wetlands, springs, floodplains and estuaries requires a precautionary approach. The policy suggests that applications for new water allocations and dams should be assessed in the context of current development, including consideration of cumulative effects on catchment resources. This consideration of cumulative effects is limited to consideration of existing allocations and setting limits for the sustainable water allocation.

Applications to take water are to consider the upstream catchment, the subcatchment to the next major user or dam or next order steam, and the catchment. Applicants are required to consider the effect on the environmental water allocation, existing users and the existing water regime. Major ecosystems dependent on the water resource are also required to be considered. Where water allocation is greater than the available yield above a dam site, further information may be required, however a license may still be provided where the overall catchment allocation remains available. Where the total available catchment allocation is taken up no water allocations are provided except where suitable evidence can be provided to demonstrate the sustainability of the proposed water extraction.

Water Resources Policy 2005/1 provides generic principles for the formulation of Water Management Plans. Among those principles are the environmental objectives of maintaining a flow regime to conserve important freshwater ecosystem values, to provide a flow regime that protects important geomorphic and ecological processes and to provide healthy refuges for instream communities during periods of low flow reflective of natural flow regimes. Historical water use that is over the licensed limit or is unlicensed is to be formalised under Water Management Plans as licenses at a lesser surety. A generic principle is included requiring the impact of the proposed irrigation to be considered in terms of the potential impact on watercourses in addition to that of the allocation. The generic principles do not address cumulative effects directly.

4.3.3 National Water Initiative

The initiative, an agreement between the States and the Commonwealth, has a number of objectives relating to national standards for water access, statutory water planning, over allocated catchments, free market trade in water, statutory provision for environmental outcomes and 'improved environmental outcomes', water use and innovation and addressing the connectivity of groundwater and surface water resources (Commonwealth of Australia 2005, 3). These objectives are to be realised through actions under the following key elements;

- Water Access Entitlements and Planning Framework;
- Water Markets and Trading;
- Best Practice Water Pricing;

- Integrated Management of Water for Environmental and Other Public Benefit Outcomes;
- Water Resource Accounting;
- Urban Water Reform;
- Knowledge and Capacity Building; and
- Community Partnerships and Adjustment (Commonwealth of Australia 2005, 4).

In terms of a planning framework, the key directive of the agreement is for the formulation of statutory water plans. A consideration of this process is that there are 'a number of land use change activities [that] have potential to intercept significant volumes of surface and/or ground water now and in the future'. The three examples given are farm dams and bores, plantation forestry and the interception and storage of overland flows. The intention of the agreement is to subject these activities to regulation and planning with their impact to be assessed 'based on an understanding of the total water cycle, the economic and environmental costs and benefits of the activities of concern, and to apply appropriate planning, management and/or regulatory measures where necessary to protect the integrity of the water access entitlements system and the achievement of environmental objectives' (Commonwealth of Australia 2005, 9)

The outcome under the agreement for the integrated management of catchments is the identification of environmental and other public benefit outcomes sought for water systems and to determine those values with as much specificity as possible. Water planning authorities are to be given the necessary resources and authority to ensure that there is sufficient water at 'the right time and place' to ensure those objectives can be met with optimal cost effectiveness'. The more detailed actions in this regard under the agreement, for example the implementation of environmental flows, are considered to be covered in Tasmania by the current principles and objectives that underpin either the *Water Management Act 1999* or Water Management Plans under the Act. One action, however, to consider 'special requirements needed for the environmental values and water management arrangements necessary to sustain high conservation value rivers, reaches and groundwater areas' is far more focused. The agreement does not directly consider cumulative effects.

The Tasmanian implementation plan under the National Water Initiative (State of Tasmania 2006) maintains that the fundamental requirements of the Initiative are covered by the current water management regime and by programs current at the time of entering into the agreement. This includes the consideration of land use activities that result in interception. One initiative under the Tasmanian implementation plan is the establishment of a project to model the impacts of plantation forestry, beginning in the Ringarooma catchment. This project is considered further in section 4.5.4. The plan also indicates that integrated catchment management is also currently achieved under the current management regime and regulation in addition to other initiatives such as CFEV, holistic environmental flows and hydrological modeling.

4.3.4 Regional Forest Agreement

The Regional Forest Agreement (RFA) is an inter-government agreement between the Commonwealth and Tasmanian governments established in 1997. The RFA attempted to delineate production forests and forests to be protected under a comprehensive, adequate and representative (CAR) reserve system (Commonwealth 1997a). The RFA emerged as a key element of the Commonwealth National Forest Policy Statement (NFPS) (Commonwealth of Australia 1995). The NFPS and the RFA applies across both private and public land. The NFPS has broad goals relating to sustainable wood production, the conservation of biological diversity and the expansion of softwood and hardwood plantations. From a catchment management perspective the NFPS goals are to ensure the 'availability of reliable, high-quality water supplies from forested land and to protect catchment values' (Commonwealth of Australia 1995, 4). While the NFPS recognises the role of vegetation cover in protecting catchments from erosion and in mitigating flood flows it does not recognise the loss of catchment yield that may occur from forestry activities. Integrated catchment management is to be achieved through the 'application of codes of practice, forest management plans and, where appropriate, land-clearing controls' (Commonwealth of Australia 1995, 25).

The RFA addresses the integrated catchment management requirements of the NFPS through the CAR reserve system, improvements to forest management systems and the establishment of a permanent native forest estate (Commonwealth of Australia 1997, 20). In Tasmania, the latter measure is implemented through the Policy for Maintaining a Permanent Native Forest Estate (Tasmanian Government 2009). This policy requires a minimum of 95% of 1996 forest area to be maintained, with thresholds provided for non-threatened vegetation communities. The policy and the criteria for the CAR reserve system (Commonwealth of Australia 1997a) do not make any explicit connection between forest reservation or distribution and integrated catchment management. The mechanism for accounting for forest clearance is through the certification of FPP's and is the responsibility of the FPA.

The recent ten year review of the RFA begins to address integrated catchment management issues in more detail and it is acknowledged in the review that this matter has received little attention over the life of the RFA (Ramsay 2008). The review recommends investment in research into the impacts of forestry practices on hydrological cycles in Tasmanian catchments, including improved models at catchment levels, improved data and catchment planning processes. These recommendations go well beyond the sustainability indicators submitted for the review by the Commonwealth and Tasmanian governments (Commonwealth and State of Tasmania 2007) which rely largely on the Forest Practices Code for management of catchment impacts. Two further recommendations begin to directly address the value of the nexus between forestry and water management planning:

- That the State ensures that its Water Management Planning framework appropriately provides for a risk-based approach to management of water interception and extraction activities in accordance with the requirements of the National Water Initiative;

- That the State requests the Forest Practices Authority to consider, in the current review of the Forest Practices Code, the inclusion of measures to enable the management of the impacts of forest practices on the yield of water in catchments, so as to meet objectives of Water Management Plans. (Ramsay 2008, recommendations 23, 24).

In response the Tasmanian government (Commonwealth of Australia 2010) committed to the development of the Water Availability and Forest Land Use Planning Tool, a forest hydrology model, which is considered further below. The response included a commitment to include the results of this modelling in water management planning for those catchments where it has been applied. The response also commits the FPA to implementing the final recommendation through the review of the Forest Practices Code which will include 'consideration of the State's water management framework commitments under the National Water Initiative, as the basis for the development of an appropriate regulatory framework to support implementation at an operation level' (Commonwealth of Australia 2010, 15).

4.4 Assessment Procedures

4.4.1 Farm Dam Approval

The ACDC is the statutory authority for the assessment and approval of farm dam applications. The ACDC is established under section 138 of the *Water Management Act 1999*. The membership of ACDC is established under the Act is to consist of two persons nominated by the Minister, a person nominated by the Minister having the administration of the *Environmental Management and Pollution Control Act 1994*, a person nominated by the Tasmanian Farmers and Graziers Association, a person nominated by the Tasmanian Chamber of Commerce and Industry and a person nominated by the Local Government Association of Tasmania (*Water Management Act 1999*, section 139). Administrative and material support is provided by DPIPWE. In addition the ACDC receives technical on any relevant environmental issues from DPIPWE.

In considering an application ACDC must seek to further the objectives of the *Water Management Act* 1999 and to act consistently with any relevant Water Management Plan. ACDC may require further information from an applicant, for example an ecological report. The enactment of the *Dam Works Legislation (Miscellaneous Amendments) Act 2007* required ACDC to consider matters normally dealt with under the *Forest Practices Act 1985* and the *Threatened Species Act 1995*. In response a set of offset guidelines were produced. These guidelines (ACDC 2007) provide a set of generic offset principles and two sets of thresholds. Where a dam proposal is determined to trigger the first set of thresholds, an offset, either informal or formal (for example a conservation covenant) is required. The second set of thresholds consist of impacts on the environment where an approval would not normally be granted unless a significant socio-economic benefit at the regional or broad community scale can be demonstrated (ACDC 2007, 13).

The offset guidelines do not specify if they are to be confined to impacts at the site only. While some thresholds, for example 'clearance and conversion of an area greater than '1 hectare' of a threatened native vegetation community' (ACDC 2007, 11) clearly relate to site specific impacts, others, such as 'adversely impacts a geographical or otherwise distinct group for an endangered species', could potentially be applied to off-site impacts. In practice, the guidelines and consideration of other matters in the assessment process are generally confined to site specific impacts. The guidelines, in addition to specific advice provided on individual proposals, form the primary reference for ACDC in considering the environmental impact of farm dam proposals. Cumulative impacts are not dealt with in the guidelines. It may be possible, for example, for a number of dams to impact upon the same threatened native vegetation community in a way that exceeds the threshold that would normally result in a single application being refused. There is no whole of catchment basis for the assessment of dams, although there is capacity within a Water Management Plan to achieve this (Davies 2001).

4.4.2 Water Allocation

The delegated authority under the *Water Management Act 1999* for the assessment of water licence applications is DPIPWE. Granting of water licences must be governed by the objectives of the Act and must conform to any relevant Water Management Plan (*Water Management Act 1999*, section 84). Beyond this the Act does not provide any guidance on how water licence applications are to be assessed. In absence of guidance through the Act, the principle guidance for assessment is provided through the Water Resources Policy 2003/1, as discussed above. The key assessment criterion is the impact on downstream users, the availability of water from the upstream catchment and the reliability of the available water resource.

The Act empowers the Minister or the delegated authority to specify conditions on any granted licence. These conditions may relate to the management of potential environmental impacts such water quality, erosion or impacts on water dependent ecosystems. Water availability may be restricted under the Act, with domestic and stock, followed by the environment, having the highest priority.

4.4.3 Forest Practices System

Where clearing of vegetation exceeds 1 ha per property per year or 100 tonnes or is intended to occur within an area classified as 'vulnerable land' under the Forest Practices Code a certified FPP is required. Exemptions from this requirement are provided in the *Forest Practices Regulations 2007* which includes dam works. In addition to requiring a FPP, a planning permit under the *Land Use Planning and Approvals Act 1993* may also be required. Private landowners may avoid this requirement through the declaration of Private Timber Reserve, the approval of which is ultimately determined by the FPA.

The Forest Practices Code provides a legally enforceable set of guidelines that FPP's must comply with (Local Government Forest Consultative Committee 2009). The FPA is the authority that has the responsibility for the regulation and compliance of FPP's. The FPA considers each FPP as it is presented, however companies, including Forestry Tasmania, that harvest in access of 100 000 tonnes each year must lodge three year projections of proposed forest practices with the FPA (Forest Practices Act 1985, section 27). Wapstra (2006) describes the Tasmanian forest practices system as operating principally at the coupe level. Biodiversity issues are considered by the Forest Practices Code and provisions are provided within it for the protection of threatened species and vegetation communities. Cumulative impacts of any kind are not considered in the assessment of a given FPP, except as they are addressed within overarching policies, strategies and regulations that may feed into the forest practices system. For example, if cumulative impacts on a given species results in listing under the *Threatened Species Protection Act 1995* then that species must be considered under prescriptions within the Forest Practices Code.

4.5 Resources

4.5.1 Trend Analysis using Historical Data

Historical data can be an important tool in determining the cumulative impact of catchment land use on catchment hydrology and water quality. In particular trend analysis of stream flow records has been used extensively to identify the impact of anthropogenic activities within a catchment. For example, Letcher et al. (2001) compared the results of a rainfall runoff model with historical stream gauge data to test predictions of land use impact on stream flow, in particular from the construction of farm dams. Nestler and Long (1997) used hydrological analysis as the basis for a cumulative impact analysis of riverine wetlands through the quantification of subtle long term changes. Similar analysis can be undertaken for water quality data.

Tasmania has an extensive network of stream gauge recording stations. The network has under gone recent expansion and there are currently 80 stations. Despite the recent additions to the network, many stations have data sets over decades that would be suitable for trend analysis of catchment land use impacts. Stations are generally sited in the lower reaches of agricultural catchments or key sub-catchments. Additional stations are operated by Hydro Tasmania. There are some gaps in the network, for example there are no current stations in the Emu or Blythe catchments despite the large area of plantations and high numbers of farm dams in those catchments. Approximately half the stations also collect basic continuous water quality data such as temperature, conductivity and turbidity.

A baseline water quality monitoring program has been implemented in Tasmania since 2003. This program, based on 53 stream gauge sites, collects basic physico-chemical water quality in addition to nutrient data derived from laboratory analysis. The sampling is on a monthly basis, however in July 2009 the sampling was reduced to quarterly and nutrient sampling ceased. A companion program, conducted on a quarterly basis, tests for the presence of common agricultural and forestry pesticides. The information from these programs has been utilised in a project to establish links between past land use, land management and water quality (Cotching and Lefroy 2007) using trend analysis. Baseline water quality data, biological data and other river health data has been shown by Dube et al (2006) to have considerable value as it can be integrated into cumulative assessment frameworks.

4.5.2 Catchment and Hydrological Models

Historical stream flow data can form the basis of catchment land use modelling. One example tested in Tasmanian catchments is the CSIRO's *TasLUCaS* model (Brown et al. 2006). This model was developed to examine the effects of afforestation, plantation development and forest age on stream flow but could be adapted to examine other land use impacts. The model predicts changes to mean annual stream flow for ungauged catchment from land use changes. For gauged catchments, the model can provide a more detailed analysis, including changes to high and low flows.

The CFEV database provides a database that can produce a CAR analysis of Tasmania's freshwater ecological values (DPIW 2008c). Although utilised as an assessment tool for individual developments, CFEV's original purpose was to indicate areas suitable for reservation as freshwater dependent ecosystems. CFEV can identify those components of the freshwater environment that are currently under reserved and have significant values determined through the CAR analysis. The need for the protection of Tasmanian freshwater values has been previously advocated by Davies (2001), while Neville (2001) advocates the application of the CAR reserve system to freshwater values in a national context. Kingsford et al. (2005) provide a framework for the holistic protection of freshwater values within a national framework that extends beyond the just the use of protected areas. Abell et al. (2007) also recognise that a CAR reserve approach also needs to be complemented by other measures specific to freshwater systems. These measures may be distant from the features of conservation value.

The use of CFEV as a planning tool for ensuring a rotational protection of catchment headwaters was recently recommended in a review of the biodiversity provisions of the Forest Practices Code (Forest Practices Authority 2009). Jerie et al. (2001) advocate a regional assessment of stream diversity in Tasmania to inform assessments, an approach that could be facilitated by CFEV. The key weakness of the CFEV database is that it is not updated and therefore some elements that are required for the analysis may become increasingly outdated.

Hydrological models for 69 priority catchments have been developed in Tasmania (DPIPWE 2010g). These models provide daily time step flow data for any point within the catchment. They can be utilised for examining the effect on the natural flow of extractions and can also produce indices of hydrological disturbance (Hydro Tasmania 2008a).

The Water Availability and Forest Planning Land Use Tool utilises the fundamentals of the TasLUCaS model, Tasmanian hydrological models and the CFEV database to allow evaluation of the impacts of changes in water availability at the sub-catchment scale on current water allocation and high conservation environmental assets (DPIW 2008f). The model was developed to assess the impact of plantation forestry development on water availability. The altered hydrological outputs were used to input into the CFEV expert rules system, which utilises a reference condition, to derived new condition indices under the various land use change predictions. There is a key limitation to the model as it was not intended to be utilised to examine small catchment issues. It was intended to 'provide guidance on water availability at a broad scale' (DPIW 2008f, 17).

4.5.3 Determination of the Impact on Yield of Farm Dams

There have been a number of methods developed in Australia to determine the impact of farm dams on catchment yield. These methods account for both ungauged catchments and where the total number and volume of farm dams may not be accurately known. A common tool used is the Tool for Estimating Farm Dam Impacts (TEDI).

Nathan et al. (2005) tested the efficacy of the assumptions used in TEDI using a more complex model, CHEAT. The results of this study found that TEDI is a practical tool for the estimation of farm dam impacts although Lowe and Nathan (2008) found that uncertainties can be significant and should be disclosed. Simulation modelling such as TEDI offers an alternative to the use of historical data with its inherent difficulties. CHEAT, however, is considered to provide a better estimate and can provide a spatially explicit representation of farm dams (Jordan et al. 2008). TEDI has been integrated into the WaterCAST catchment model to assess farm dam impacts on streamflow at the catchment level. This model allows for variables such as land use to be considered in an understanding of catchment response to alterations in stream flow (Cetin et al. 2009).

There are generic types of information required for any quantitative assessment of farm dam impacts using models such as TEDI or CHEAT. Volume, size distribution, magnitude of demands and seasonal pattern are all key variables in such an assessment. Lowe et al. (2005) used engineering plans and a remote sensing derived digital elevation model to develop a relationship between dam surface area and volume. Dam numbers and surface areas were determined through the use of aerial photography. McMurray (2004) presents a method for utilising geometric relationships to estimate volume for dams at a sub-catchment level. Jordan et al. (2008) used satellite imagery to achieve estimates of farm dam volume, an approach also utilised in a number of other studies (Murray Darling Basin Commission 2008, Dare et al. 2002, Department of Sustainability and Environment 2004) often in conjunction with aerial photography and topographic data. Although the analysis of the digital photography was automated, each waterbody identified was manually checked to determine if the feature was a farm dam. Landowner surveys in different regions were utilised to estimate average demand factors. The demand factor is the annual average water usage divided by the dam volume. These parameters can be fed directly into a simulation model such as TEDI. In the absence of the use of such a model Lowe et al. (2005) derived regional prediction equations for dam impacts based on applying TEDI to gauged catchments.

4.6 Summary

A sound argument can be made that the principles and objectives of the RMPS must require consideration of impact cumulative effects arising from the utilisation of natural resources. It is difficult to imagine how the sustainable use of Tasmania's resources, the principle aim of the RMPS, can be achieved without such consideration. Without explicit reference in the RMPS, however, their consideration is not certain, as is evident from the regulatory processes governing farm dam construction. The experience from jurisdictions where cumulative effects are explicitly dealt with under legislation indicates the value of that approach in forcing consideration of cumulative effects.

Should the RMPS principles and objectives include an explicit reference to the need to consider cumulative effects, the experience in USA, Canada and the European Union suggests that such a directive requires a supporting set of guidelines. State Policies offer an opportunity to provide such a guideline at a sufficiently overarching regulatory level but also with sufficient detail. The development and application of State Policies would require greater emphasis than is currently evident for this measure to be effective. To date only three State Policies have been formulated.

The absence of an explicit reference to cumulative effects in the RMPS flows on to the *Water Management Act 1999* where cumulative effects are also not directly considered. This Act provides little guidance in the assessment of environmental impacts other than high level general principles. These are open to interpretation and are potentially oppositional and are not ordered in a hierarchy of priority. The policies that have therefore been required also provide broad principles relating to protection of environmental values but do not identify cumulative effects as a significant matter for consideration. Both the Act and associated policies have an emphasis on water allocation and yield as the determinate both of provisions for the environment and for utilisation of the water resources. This is also largely reflected in the formulation of Water Management Plans. In this sense cumulative effects are dealt with in water management planning through the recognition that individual proposals, with some exceptions under the Act, to extract a portion of the water resource are assessed against the overall utilisation. This approach, however, does not consider, particularly in relation to farm dams, impacts other than reduction in yield. Furthermore, the spatial scale is generally at the catchment level and actual impacts, such as farm dam usage, may be not be accurately accounted.

The forest practices system in Tasmania operates largely independently of other natural resources management processes, including water resource management. It is not encompassed by the RMPS. Again, while 'sustainable management' is an objective of the forest practices system, there is little guidance on how this is to be achieved in a strategic sense and cumulative effects are not explicitly addressed.

Where strategic issues have been addressed, such as through the RFA, they have a terrestrial, forest community and forest species focus. The Forest Practices Code, the key regulatory guidance for forest practices, is very much a site specific guide for the formulation of individual FPPs.

More recent developments have begun to recognise the need for integrating forest practices into water management planning. Both the NWI and the recent review of the RFA recognise the need to achieve this outcome. The recent review of the biodiversity provisions of the Forest Practices Code recommends the CFEV catchment spatial scale as one tier of strategic management (Biodiversity Review Panel 2008). However, the result of these developments has largely been to rely on the current water resource planning approach, particularly through the development of Water Management Plans, and to link water management planning to the forest practices system through the Forest Practices Code. While there is a capacity within the *Water Management Act 1999* and Water Management Plans for an integrated catchment management approach, the weakness of the links to the forest practices system raises questions regarding the effectiveness of attempting a comprehensive integrated catchment management approach within the current framework.

Having catchment management legislation as the primary legislation for all activities within a catchment must be a prerequisite for the integrated management of catchments that accounts for other interception activities such as forestry as it would overcome the issue of conflicting sectors. In addition this would also be more effective in taking a strategic approach to the management of other issues such as impacts on headwater streams, sediment loads and reservation of freshwater systems.

The development of hydrological and catchment land use models in combination with the CFEV database provide an increasing opportunity to address broad catchment management issues in an integrated approach. These tools also allow for a quantitative assessment of future trends in land use and water management. The importance of long term stream flow, water quality and biological monitoring in the development of these models and the assessment the impact of past land use trends is evident. The continuation of these programs is vital as future applications of the data may yet to be recognised. In this context the decision to downgrade the baseline water quality monitoring in 2009 would seem short sighted. The CFEV database offers a powerful tool for the strategic management of catchment values and the implementation of the CAR reserve system for freshwater ecosystems but it should be sufficiently resourced to allow it to be based on the latest relevant information.

The primary knowledge gap for the conduct of CEA in the context of Tasmanian catchments is the need for more refined tools to examine impacts at smaller spatial scales, particularly within headwater streams and sub-catchments with intensive development. One of the key requirements is to accurately account for current usage For farm dams this requires an accurate account of farm dam location, area, volume and seasonal usage. For water licenses, metered extraction points should be a priority. While there are significant resources available for the CEA in Tasmania, a more direct regulatory requirement of CEA would allow for the development of monitoring programs and tools specifically designed for examination of cumulative effects. One key area, for example, is the need to increase knowledge of special values such as burrowing crayfish and to further knowledge of the ecology of headwater streams.

Chapter 5 Discussion

5.1 The Imperative for Considering Cumulative Effects

At least since the 1970's there has been an awareness of the potential for multiple stressors on the environment to have a significant cumulative impact in combination. This awareness developed into an impetus for appropriate legislation, initially in the United States and subsequently in other jurisdictions. Cumulative effects on the environment are now required by law to be addressed in environmental assessments in the United States, Canada and the European Union for a number of decades. As a result a considerable body of work has been produced examining appropriate methods, approaches and frameworks for the practice of CEA. CEA from these jurisdictions provides substantial evidence that cumulative effects can be key drivers of significant environmental impacts.

The review of the literature leaves no doubt regarding the importance of CEA. While there are inherent difficulties recognised with the regulation and practice of CEA, there is no body of opinion within the literature that argues that CEA is unnecessary. There is a consensus that without adequate consideration of cumulative effects it is not possible to properly determine if natural resources are being managed sustainably.

5.2 Legislation and Evolving Practice

The requirement of federal legislation in the United States and Canada, and analogous legislation in the European Union, is the fundamental trigger for the breadth and extent of the assessment of cumulative effects in those jurisdictions. The requirement to determine cumulative effects that may arise from the implementation of plans, programs and projects, particularly those from government, is an important component of those regulatory mechanisms. The legal testing of the adequacy of environmental assessments in the Unites States has focussed on how cumulative effects were addressed as required under federal legislation. This indicates the power of legislation to provide an impetus for the proper consideration of cumulative effects. It is unlikely that cumulative effects would be addressed effectively in environmental assessment otherwise.

Despite the power of the law it is evident that a legal requirement does not guarantee that effective practice to further a regulatory aim will necessarily occur. The level of rigour required if CEA practice is to adequately meet the intention of regulation is often not being realised. This is despite the principals and fundamentals of CEA being considered to be well established as are a number of practical tools and methods. Issues such as poor understanding of principles and methods cannot be said to be wholly unique to CEA practice, although it may be reflective to some degree of its evolving nature. A key issue in the practice of CEA is its inherent complexity and the difficulty of addressing that complexity within the constraints of project EIA. Another key issue from a regulatory and management perspective is that assessments at the project EIA level are reactive. The process is triggered by a proposal entering the regulatory system may offer limited opportunity for any substantial changes.

In this context, the growing identification of the role of SEA in the assessment of cumulative effects has some promise. The greater scope of SEA is a key advantage. It allows for a better capture of all the relevant impacts or proposals across space and time. SEA, although it allows for a higher level of assessment that provides for greater control over the direction of development, still requires a triggering action. In that sense it is still largely a reactive process. There are a number of issues relating to the effective use of SEA to consider cumulative effects. The evolving nature of SEA itself, inherent difficulties within its application and methodological challenges in integrating CEA within SEA suggest that considerable work is required. The promise and challenge of this approach is to establish a feedback mechanism or 'tiering' from SEA into project EIA where SEA sets the context of EIA while the results of individual EIA's feed into the strategic approach.

The approach of regional planning offers a more pro-active approach to managing cumulative effects. Such an approach would define resource limits which account for current and predicted cumulative effects. In this way guidance on development both spatially and temporally is offered. This would allow for individual proposals to be measured within a strategic framework, providing for a better assessment of cumulative effects at the project level. A key weakness in this approach, which would require partnerships across governments and other sectors, is that without a legislative trigger it would depend on other, possibly weaker, mechanisms.

5.3 Cumulative Impacts of Farm Dams, Forest Practices and Water Licenses

5.3.1 Farm Dams

The review of the literature examining catchment impacts of farm dams, water allocations and forest practices has shown that cumulative effects from these activities have the potential to adversely impact on the environment.

In Australia there has been an increasing attention given to the cumulative impact of farm dams on catchment yield. That work, driven by water allocation issues, has shown that farm dam capacity has a direct relationship to reductions in catchment yield. That work also identifies that the impact is better considered in terms of the natural flow regime. The impact on the natural flow regime from farm dams is shown to vary spatially and temporally. Furthermore, there are significant unknowns. Usage, total farm dam numbers and storage, including unlicensed and illegal dams, all require quantification in order to understand the actual impacts of farm dams.

Consideration of additional farm dam impacts such as water quality impacts, fragmentation and habitat alteration are poorly considered in the Australian literature. Research from the northern hemisphere suggests that these impacts may be significant and require further investigation at the catchment scale. It is reasonable to presume that those impacts are also likely to occur in some form in Australian and Tasmanian catchments and there is a need for further study of these impacts in the Australian context.

5.3.2 Forest Practices

The impact of forest practices on catchment yield has also been the subject of increasing attention in Australia, a reflection of both the increase in plantation area and the increasing scarcity of the water resource in Australia. The review of the associated literature indicates that the linkages between vegetation change, forest age and runoff are well understood. Other potential impacts such as increased sediment loads are also well documented. Methods exist for the estimation within the catchment scale of these impacts. What is not well quantified is how these effects interact cumulatively with other water interception activities and land use impacts, in particular the downstream extent of impacts within catchment headwaters.

Forestry is often concentrated in the upper catchment. Research from Tasmania has established a measurable impact of forestry activities on freshwater biota and processes within headwater streams. There are two challenges arising from this work. The ecology of headwater streams requires further study in order to further understand how processes and biota might be impacted by forest practices. Secondly, an understanding is required of how these results can be extended across upper catchments in a meaningful way that can underpin management.

5.3.3 Water Extraction

The impact of water extractions on the natural flow regime is perhaps the most understood of the three impacts examined. The literature examining the links between flow and physical and biotic processes is considerable. This is not to suggest, however, that this understanding is in anyway complete. Research continues to reveal and refine our understanding of flow and freshwater ecosystems. What is evident is that these relationships are complex and alterations in the natural flow regime may have consequences that are not readily appreciated. All the variability in flow within a river or wetlands system is of importance. These relationships also need to be established for individual catchments.

5.4 *The Tasmanian Context*

5.4.1 Great Forester – Brid Catchment Case Study

Consideration of farm dams, water licenses and forest practices within the Great Forester – Brid catchment has shown that the potential for the cumulative impact of these activities to be significant is high. While the approach taken in considering those cumulative effects can be considered to be a first pass or scoping analysis, it has revealed measurable cumulative impacts at that level of investigation. Impacts on the natural flow regime, connectivity, on special values and high conservation priority reaches are evident in this context. The catchment case study identified that temporal and spatial scoping is an important consideration, as identified in the cumulative effects literature. It is evident that cumulative effects within this catchment must be considered at least at the sub-catchment scale, across catchments for some values and at least at monthly time scales.

The catchment case study also identified a number of issues for the management of cumulative effects. Despite impacts on yield from water extraction and farm dams being the simplest impact to consider and the subject of the greatest regulatory focus, there are considerable knowledge gaps. The extent of all farm dams in the catchment is not accurately known and critically the actual usage is also unknown for both farm dams and water extraction. Without this knowledge the impacts of these activities, particularly at appropriate spatial and temporal scales, cannot be accurately determined. The other impacts examined were analysed at a high level and predominately in terms of spatial extent. The actual impacts of fragmentation by farm dams or forest practices in the catchment headwaters requires more detailed investigation founded on an appropriate monitoring methodology. An attempt was made to consider impacts of farm dams on vegetation communities and threatened flora, however this information is not collected adequately to allow for such an assessment. A review of the types of information collected in assessments of farm dams is required for the full range of cumulative effects to be better quantified.

In the Great Forester – Brid catchment the combined number of individual regulatory decisions for these activities is cause for concern and is certain to increase into the future. The level of development within the catchment is not unique. There are other Tasmanian catchments with a greater extent of plantations and forest harvesting, the catchments of northern Tasmanian nearly all have greater farm dam densities and most of these also have greater levels of irrigation. Other catchments may have less intensive use of one or more of these activities but have a more varied and unpredictable flow regime. The proposed irrigation development in the catchment is significant and similar proposals are planned for other catchments in the region and across Australia. The literature identifies that all these farm dams, forest practices and water licenses have in-direct, additive and synergistic effects on the environment. The central question of the exact nature of the cumulative effects of these activities in combination in the Great Forester – Brid catchment and other Tasmanian catchments is currently unknown.

5.4.2 Regulation

In the RMPS Tasmania has an overarching set of principles to guide the regulation of natural resources. While considered to be a promising model, the degree to which the objectives are applied within the day to day regulatory decision making in Tasmania is doubtful. The impact of large development proposals may be meaningfully tested against the objectives, however it is difficult to see how a proposal at the scale of a farm dam can be properly assessed against those same objectives. Such a proposal is likely to be seen to have negligible impact or to have an impact that can be offset through protection of similar values elsewhere. In this sense for an individual proposal the objectives can be seen as being met. It is in this way that cumulative effects occur within the Tasmanian regulatory system. Without an explicit requirement to consider cumulative effects, and without an associated framework within which to assess them, the objective of the RMPS to achieve the sustainable use of Tasmania's natural resources cannot be met.

The emphasis in much of the CEA literature is the consideration of multiple impacts of developments of moderate to large scales, for example hydro electric and other large impoundments, pulp mills and mining operations. Although the difficulties of considering cumulative effects at the project EIA for this scale of proposal are recognised, these types of proposals would normally expected to be supported by sufficient resources to conduct an effective CEA component of a project EIA. In Tasmania it has been shown that smaller, more numerous regulatory decisions within an environment where cumulative effects are not addressed have the potential to result in significant environmental impacts. In Tasmania, farm dams, water licenses and, on private land, FPP's are normally sought by private individuals or small scale private enterprises. It is not reasonable to require these applicants to consider cumulative effects across appropriate temporal and spatial scales in support of proposals that are often limited in size and capital investment and without guidance. In the same way, regulatory decision makers such as the ACDC cannot be expected to consider cumulative effects without some form of overarching guidance or cumulative assessment framework.

A regional, pro-active approach that considers the full extent of all cumulative impacts may provide such a framework. This approach can consider the past and potential cumulative impact on selected VEC's of a variety of activities. By setting limits on the impact to those VEC's, decisions can be made between competing interests using a common reference. The greater benefit can then be determined between different activities. For example, all activities in a particular sub-catchment can be assessed equally in terms of impact on a number of VEC's, such as the natural flow regime, water quality or a threatened aquatic species. In this way the environmental impact and economic benefit of each activity can be assessed against development limits and priorities determined for the sub-catchment. Individual proposals within the sub-catchment can then address cumulative effects by providing a case for the benefit of the proposal against the 'usage' of a particular development limit. It may also provide motivation for proponents to move beyond a property fence line view towards a more strategic approach using shared resources.

Cumulative effects are currently indirectly dealt with to some degree within current water management planning and Water Management Plans through the allocation of environmental flows and the assessment of water availability for water licenses. To date this management has had a strong emphasis on broad assessments of the available water yields at the catchment level and has not considered in any way other impacts such as fragmentation through farm dam construction or land use impacts. Attempts to link forest practices within a catchment to water management planning are yet to be realised. While the forest practices system remains outside of the RMPS these linkages are likely to be weak, as is the use of the Forest Practices Code to protect freshwater dependent ecosystems.

Recent reviews have identified the need to plan forestry activities in a strategic way to protect freshwater values. Any strategic approach in this regard would be better integrated into an overall integrated catchment management approach to the protection of those values as there is an inherent difficulty in impelling one sector to account for cumulative effects that either impact on, or occur through, other sector activities. The key to achieve integration would be to establish a principle catchment management legislation that would have primacy over other resource management legislation within a catchment. This would provide an administrative boundary to overlap the physical catchment boundary, thereby facilitating a key requirement for the determination of the spatial boundaries of CEA assessment. The forest practices system in particular, but also arguably current water management planning, can be thought of as having a narrow focus and therefore meeting the description provided by Conacher and Conacher (2000, 13) of 'resource management'. Integrated catchment management would allow both systems to evolve into a broader, holistic approach described by the same authors as 'environmental management'.

Chapter 6 Conclusion

The consideration of cumulative effects has long been recognised as a fundamental aspect of ensuring sustainable use of natural resources. Equally it is understood that environmental assessment at the level of individual proposals fails to adequately address the issue of cumulative effects in a meaningful and effective way. This holds true for proposals across the spectrum of perceived significance, however, the situation is exacerbated within regulatory frameworks that make numerous and ongoing assessment decisions for small proposals without requiring the consideration of cumulative effects. In Tasmania, the regulation of the construction of farm dams, allocation of water licenses and the preparation of Forest Practices Plans fall into that category.

The RMPS is the logical instrument for the explicit requirement in law for the consideration of cumulative effects in Tasmanian. There are, however, a number of key pieces of natural resource legislation, for example the *Forest Practices Act 1985*, that are not under the umbrella of the RMPS. Without the RMPS applying to all relevant legislation its effectiveness in triggering consideration of cumulative effects in Tasmania would be compromised. It is imperative, therefore, that the RMPS encompass all relevant natural resource legislation.

As seen from the experience in the United States, Canada and elsewhere, legislative triggers alone do not provide sufficient guidance nor do they guarantee effective practice. The *State Policies and Projects Act 1993* provides a suitable vehicle for establishing an enforceable mechanism at a high level that would provide a framework for the assessment of cumulative effects. Furthermore this approach could also provide the legislative impetus for the development of pre-emptive regional strategies that properly account for the impact of cumulative effects on the environment and provide for development guidance in this regard. The formulation of a State Policy could provide the linkages between strategic or regional assessment and project assessment, and the publication of guidance for those assessments. There is no doubting the difficulties in formulating such a policy, however the principal impediment would appear to be the provision of sufficient resources and a will to use this mechanism. Currently State Policies are under utilised; they have been slow in formulation and are few in number.

An important element of a strategic regional approach should be the implementation of a CAR reserve system for freshwater systems. Tasmania has a diverse range of aquatic ecosystems, which include numerous endemic species, and the protection of these values should be a priority in the same way as protection of terrestrial ecosystems. The establishment of a CAR reserve system could be applied in an integrated way with development limits set out in a regional framework. In some instances, for example headwater streams, total protection extending to the entire catchment may be required to establish areas where natural processes are undisturbed. These areas would also provide valuable ecosystem services to downstream reaches that may mitigate the impacts of current downstream development or possibly allow for an expansion of development limits. Other areas could be protected through limits to upstream impacts, for example water quality or the natural flow regime. Fragmentation is one impact that would drive both the selection of reserves and the establishment of limits to development impacts within particular sub-catchments.

Integrated catchment management would form an important component of any regional approach to cumulative impacts. Integrated catchment management is the appropriate management approach that properly encompasses all the activities that may have an impact on catchment ecosystems and processes. It encompasses the full range of environmental concerns, linking water and land management. The inclusion of CEA explicitly within integrated catchment management planning constitutes best practice legislative basis for river management and would avoid the 'tyranny of small decisions' evident in the case study of the Great Forester – Brid catchment.

The following recommendations are made for the implementation of CEA in Tasmania with particular reference to freshwater dependent ecosystems and processes at the catchment spatial scale:

- Consideration of cumulative effects in natural resource management and assessment of development proposals must be an explicit requirement of the RMPS. This requirement should extent to all sectors, including government, and should apply to plans, programs and policies in addition to individual development proposals. All legislation regulating natural resource management and planning should conform to the RMPS.
- A State Policy under the *State Policy and Projects Act 1993* is an appropriate mechanism for providing more detailed regulatory guidance for the assessment of cumulative effects.
- Integrated catchment management through the implementation of primary catchment management legislation is a necessary component for the adequate management of cumulative effects at the catchment level. All other natural resource management legislation, including those that govern forest practices, should be required to conform to catchment management regulation.
- Catchment management must include management of the full range of cumulative and direct impacts from all activities within the catchment. This management must be at a spatial and temporal scale that is appropriate to achieve this.
- Integrated catchment management should be one component of a pro-active regional approach to natural resource management that sets development limits through the consideration of direct and cumulative impacts on selected VEC's. This approach allows proponents of small to medium scale proposals and assessment authorities to consider cumulative impacts within a strategic framework.
- Integrated catchment must include the provision of freshwater reserves based on the CAR principles.

- Whilst current resources allow for the initiation of these recommendations, further refining of monitoring, research and modeling efforts is required. In particular, knowledge of freshwater dependent VEC's needs to be increased. Further research into headwater stream ecology and processes is required. Refining of the spatial scale of catchment models to allow for consideration of sub-catchments is required. The most pressing need is to accurately account for current impacts, in particular actual water extraction rates and dam usage rates, accurate positioning of dams in the landscape and their area and volume.

The absence of any meaningful consideration of cumulative effects has resulted in the potential for significant impacts to freshwater ecosystems and processes to occur across and within catchments in Tasmania. These impacts will continue unless appropriate legislation, policies and strategic frameworks are implemented and appropriate methods for CEA in Tasmania developed. It is acknowledged that considerable resources may required and that the issues are complex. However, given that only one aspect of the regulation of natural resources in Tasmania was examined here, and that unregulated activities are likely to contribute at least as much to cumulative environmental impacts, the adoption of CEA in Tasmania is seen as essential.

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